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This document provides the technical background and justification for the U.S. Environmental Protection Agency's (EPA's) proposed regulation (40 CFR Part 503) covering the surface disposal (impoundments, lagoons, waste piles, etc.) of municipal sewage sludge. Because a risk assessment methodology for surface disposal has not yet been developed, the document presents the risk assessment methodology used to derive risk-based limits for contaminants in monofilled sewage sludge, which can be modified to derive risk-based contaminant limits in surface-disposed sludge. The underlying assumptions of the methodology, embodied in a series of computer models, are discussed, and the values assigned to the model parameters for monofills are presented and justified. The maximum allowable sludge contaminant concentrations for surface-disposed sludge (based on 98th percentile concentrations in a sludge quality survey) are then presented, and the results of a sensitivity analysis for monofills are also included. Finally, the various siting limitations and management practices mandated by the regulation are discussed and justified.

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TECHNICAL SUPPORT DOCUMENT

Surface Disposal of Sewage Sludge

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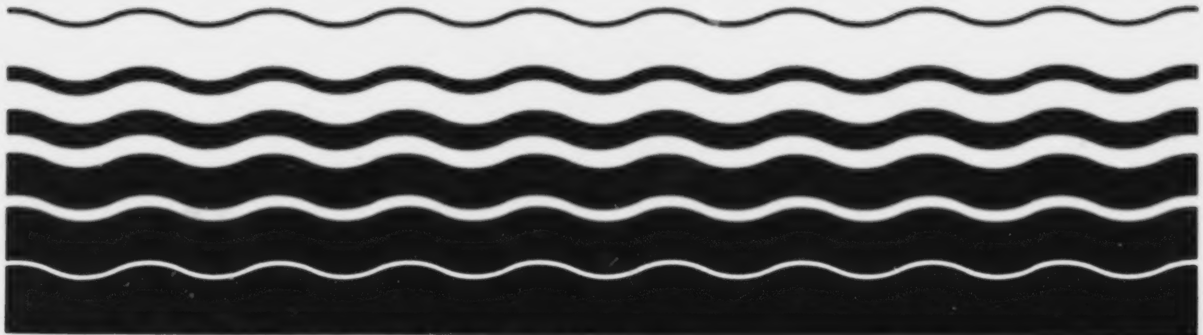

PREFACE

Section 405(d) of the Clean Water Act requires the U.S. Environmental Protection Agency (EPA) to develop and issue regulations that identify:

- Uses for sludge, including various means of disposal
- Factors, including costs, which must be considered when determining the measures and practices applicable to each use or disposal method
- Pollutant concentrations that interfere with each use or disposal method

To comply with this statutory mandate, EPA has embarked on a program to develop five major technical regulations for the following areas: land application/distribution and marketing; monofilling; surface disposal; incineration; and reduction of pathogens and vector attraction. EPA also has proposed regulations governing the establishment of State sludge management programs, which will implement both existing and future criteria (40 CFR 501).

The primary goal of the proposed surface disposal regulation is to protect human health and the environment. Included in this proposed regulation are provisions to protect air and groundwater quality from the deleterious effects that may occur due to sludge surface disposal. This document provides the technical background and justification for the provisions contained in Subpart E of the proposed regulation.

Public comment on the technical adequacy and scientific validity of this document as well as on the requirements contained in the proposed regulation should be submitted during the public comment period. Any questions related to this document may be directed to:

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LIST OF UNITS AND ACRONYMS

ac	acre
BEHP	bis(2-ethylhexyl)phthalate
bw	body weight
CFR	Code of Federal Regulations
CWA	Clean Water Act
DDD	Bis 1,1-(4-chlorophenyl)-2,2-dichloroethane
DDE	Bis 1,1-(4-chlorophenyl)-2,2-dichloroethylene
DDT	Bis 1,1-(4-chlorophenyl)-2,2,2-trichloroethylene
Eh	oxidation-reduction potential
EO	Executive Order
EPA	U.S. Environmental Protection Agency
FAA	Federal Aviation Administration
FR	Federal Register
ft	feet
ha	hectare
HEL	health effects level
hr	hour
kg	kilogram
k_d	distribution coefficient
K_{oc}	organic carbon partition coefficient
K_{ow}	octanol-water partition coefficient
K_{sat}	saturated hydraulic conductivity
L	liter
L_c	leachate concentration
L_{cl}	limiting leachate concentration
LEL	lower explosive level
ln	natural logarithm
m	meter
m^2	square meter
m^3	cubic meter
MCL	maximum contaminant level
MEI	most exposed individual
MF	modifying factor
mg	milligram
mL	milliliter
mm	millimeter
mph	miles per hour
MPN	most probable number
mt	metric ton
mv	millivolt

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LIST OF UNITS AND ACRONYMS (Continued)

NOAEL	no-observed-adverse-effects level
OHEA	Office of Health and Environmental Assessment
ORD	Office of Research and Development
OWRS	Office of Water Regulations and Standards
PCB	polychlorinated biphenyl
pH	relative hydrogen ion concentration
RCRA	Resource Conservation and Recovery Act
RfD	Risk Reference Dose
SAB	Science Advisory Board
S_c	sludge concentration
S_d	limiting sludge concentration
su	standard units
$t_{1/2}$	environmental half-life
t_c	contaminant travel time
TCE	trichloroethylene
TKN	total Kjeldahl nitrogen
TOT	time of travel
TW	water travel time
ug	microgram
UF	uncertainty factor
WRC	Water Resources Council
yr	year

SECTION ONE

TECHNOLOGY AND PROCESSES

A number of different methods have been developed to reuse or dispose of sludge by application to land. Monofilling and land application (including agricultural and nonagricultural applications and distribution and marketing) have been considered in the regulation of sewage sludge under Part 503. In addition, a surface application practice termed surface impoundment is used as a treatment method or for interim storage. Yet another current practice is to dispose of sewage sludge by placing it in "piles" on land surface. This method is similar to other land application practices but is not strictly covered by those terms. The Agency is proposing to call the use of surface impoundments and waste piles sewage sludge surface disposal. A sewage sludge surface disposal site is a dedicated area of land on which sewage sludge is disposed and on which the sewage sludge remains for 1 year (yr) or longer.

In 1984, when the Agency initiated the Part 503 rulemaking process, surface disposal sites were defined as treatment or interim storage facilities. Later the Agency learned that some communities use surface disposal sites for extended periods of time, a practice which suggests that surface disposal can be used for final disposal of sludge. Because of this practice, surface disposal sites must be regulated under the Clean Water Act (CWA).

Sewage sludge surface disposal is similar to a number of other methods of surface application. For example, it is similar to the nonagricultural land application known as spreading sludge on dedicated land. With a sewage sludge surface disposal unit, however, no daily or final cover is established on the sewage sludge (such a cover is required on dedicated land). Furthermore, a 1-yr time delay occurs before the area of land containing the surface sludge is defined as a surface disposal unit.

A sewage sludge surface disposal unit in which sludge is placed on land that is surrounded by a containment wall, in a natural topographic depression, or in a man-made excavation resembles a sewage sludge monofill. Similar to dedicated land, however, a daily or final cover is required for a monofill but not for surface disposal. Additionally, monofills are defined as disposal from the moment sludge is first placed in them, whereas surface disposal is defined to occur only after the sludge has been stored for over 1 yr.

Sludge may be placed on land that is surrounded by a containment wall, in a natural topographic depression, or in a man-made excavation for treatment as well as for disposal. A surface disposal unit, therefore, may resemble treatment facilities such as pits, ponds, and lagoons. The definition of surface disposal, however, is intended to cover ultimate disposal, not treatment, of sludge. The Part 503 requirements cover only final use or disposal practices.

The Agency has no current data on the practice of disposing sewage sludge at sewage sludge surface disposal sites; however, the Agency does have data on surface impoundments, which are considered part of the wastewater treatment train. This information is relevant to surface disposal because when an impoundment is abandoned as a treatment facility it becomes a surface disposal site if the sludge is not removed to another disposal site within a year after the treatment processes cease.

In the 1986 census of Subtitle D [Resource Conservation and Recovery Act (RCRA)] facilities (EPA 1988c) identified 1,938 surface impoundments that contain municipal sewage sludge. These impoundments constitute approximately 1% of the 191,500 nonhazardous solid waste surface impoundments identified in the census. The census defined a surface impoundment as:

> A part of an establishment which is a natural topographic depression, man-made excavation, or diked area formed primarily of earthen materials (although it may be lined with man-made materials) that is designed to hold an accumulation of liquid wastes or wastes containment free liquids. Treatment, storage, and disposal surface impoundments are included.

The following information has also been gathered on surface impoundments, as defined by RCRA:

- Most impoundments are small. Eighty-five percent of those surveyed are smaller than 1 acre (ac). Ninety-six percent receive 50 or fewer gallons per day.
- Thirty percent of Subtitle D surveyed facilities had synthetic or clay liners.
- Thirty percent had overtopping controls.
- Few facilities have monitoring systems. Only 6.8% have groundwater monitoring, 2.6% have surface water monitoring, and 0.5% have air emissions monitoring.

In addition to treatment, sludge lagoons provide a low-cost method for sludge dewatering. To reduce odor problems, sludge is typically stabilized before being placed in a drying lagoon. In some cases, however, sludge lagoons are used to stabilize raw sludge as well as dry the sludge.

Nationwide, many smaller publicly owned treatment works (POTWs) place sludge in lagoons, a practice that is particularly common in the Southeast and Southwest. Some larger POTWs use sludge lagoons as well. Some of the approximately 2,000 sludge lagoons in use throughout the United States are emptied and replenished infrequently. Lagoons that were once part of a wastewater treatment train are at times bypassed when they become filled with sludge and remain undisturbed for years. These lagoons are one kind of surface disposal site that this document addresses.

In other cases, according to the Agency, the sludge is simply piled on the ground or left within a containment. No attempt is made to add vegetative cover or otherwise manage the surface disposal site.

Information about the number of domestic sewage sludge surface disposal sites, their size and design, typical management practices, and the amount of sewage sludge disposed in or on them is being solicited in the preamble to the proposed Part 503 regulations. According to the Agency, the practice of surface disposal is limited and sites are relatively small.

SECTION TWO

RISK ASSESSMENT METHODOLOGY

2.1 INTRODUCTION

The Environmental Criteria and Assessment Office in the Agency's Office of Research and Development (ORD) has developed detailed risk assessment methodologies for three sludge disposal options: land application/ distribution and marketing, landfilling, and incineration. The first step in developing these risk assessment methodologies was to identify the major pathways of exposure through which each sludge contaminant could reach and harm humans, plants, or animals. Mathematical expressions (algorithms) were then developed to describe the transport, fate, and effects of the pollutants in each medium (air, water, soil). Site-specific factors that influence exposure and toxicity, such as soil or climatic variability and management practices, were also incorporated in the algorithms. The risk assessment methodologies were peer-reviewed internally by the ORD and externally by the Agency's Science Advisory Board (SAB) and then translated into computerized mathematical models by the Office of Water Regulations and Standards (OWRS).

The risk-based computer models developed from the risk assessment methodologies predict the movement of pollutants from a source (e.g., sludge placed on or in the land, emissions from an incinerator) through several environmental pathways, compartments, or media to reach a maximum-exposed individual (MEI) or nonhuman target organism at a calculated concentration. The calculated pollutant concentration is then compared to either an environmental impact criterion or human health criterion. The criteria were derived from different sources, depending on the type of pollutant:

- For noncarcinogens, a human health criterion is the daily intake of pollutant that will cause no adverse effect. The models use the

Agency's established risk reference doses (RfDs) for humans as the threshold values for chronic toxicity.

- For plants and animals, the models use chronic toxicity threshold values as reported in the scientific literature.
- For carcinogens, the only daily intake level of a pollutant that produces no risk is zero. The Agency's Carcinogen Assessment Group has estimated the carcinogenic potency (i.e., the slope of risk versus exposure) for humans exposed to low-dose levels of carcinogens (EPA, 1987). These potency values are used in the models to derive the exposure level expected to correspond to a given level of excess risk. Incremental risk levels of 10^{-6} (one additional cancer case in one million people) to 10^{-4} (one additional cancer case in ten thousand people) were evaluated for this regulation.
- Maximum contaminant levels (MCLs) established by EPA's Office of Drinking Water (ODW) are used as the drinking water criteria for the ground water contamination pathway evaluated for landfilling and land application.
- The seven ground water pollutants for which no MCLs have been proposed are regulated with the target concentrations carcinogenic potency (q_1*) values associated with specific risk levels for the MEI or with an RfD.

The risk-based model for each disposal practice is used to back-calculate the maximum pollutant concentration in sludge disposal rate, discharge rate, or emissions level that will yield the RfD, MCL, or appropriate carcinogenic dose at the exposure point for the MEI or target organism. The MEI is defined differently for each disposal practice. In general, the MEI is an adult with a 70-yr exposure to long-term average contaminant levels from a sludge use or disposal practice in close proximity to his residence.

No risk assessment methodology has been developed for surface disposal. ORD is now developing such a methodology, which will be proposed in 1989. This methodology and the associated risk-based computer models are expected to parallel closely the methodology and models developed for the monofilling disposal option. The set of computer models that incorporates the monofill methodology, collectively known as SLUDGEMAN, is described in the sections below. SLUDGEMAN is designed to calculate a reasonable worst-case level of exposure to chemicals leaching from a landfill and emitted to the atmosphere. Details of the model, which can estimate the risk associated with sludge disposal in a given landfill, are provided here as an approximation of the surface disposal model.

2.2 IDENTIFICATION OF PATHWAYS OF EXPOSURE FOR SLUDGEMAN

SLUDGEMAN predicts the movement of pollutants from a source landfill through several environmental pathways to reach a nonhuman target organism or an MEI defined as an adult with a 70-yr exposure to long-term average contaminant levels released from a landfill near his or her residence, at a calculated concentration. This calculated pollutant concentration is compared to either an environmental impact criterion or a human health criterion to determine the risk of disease to the MEI. This section describes the key pathways of exposure that were identified during the development process.

Four environmental pathways were identified as critical to the analysis of sludge monofills (these same pathways are relevant to surface disposal):

- Contaminant infiltration to ground water
- Vapor loss from fill material
- Surface runoff
- Suspension of contaminated particles from the working face

Figures 2-1 to 2-3 show routes of human exposure from landfilled sludge and contaminant migration pathways for narrow- and wide-trench landfills. The first two pathways were identified as the key pathways of contamination and were modeled in EPA's SLUDGEMAN model.

Infiltration of sludge contaminants to ground water and subsequent ingestion of this water by humans was considered the most significant of the potential pathways, based on the likelihood of occurrence. The MEI for this groundwater pathway was defined as an adult residing at the property boundary of the landfill who drinks 2 liters (L) of the ground water every day for his or her entire 70-yr lifespan.

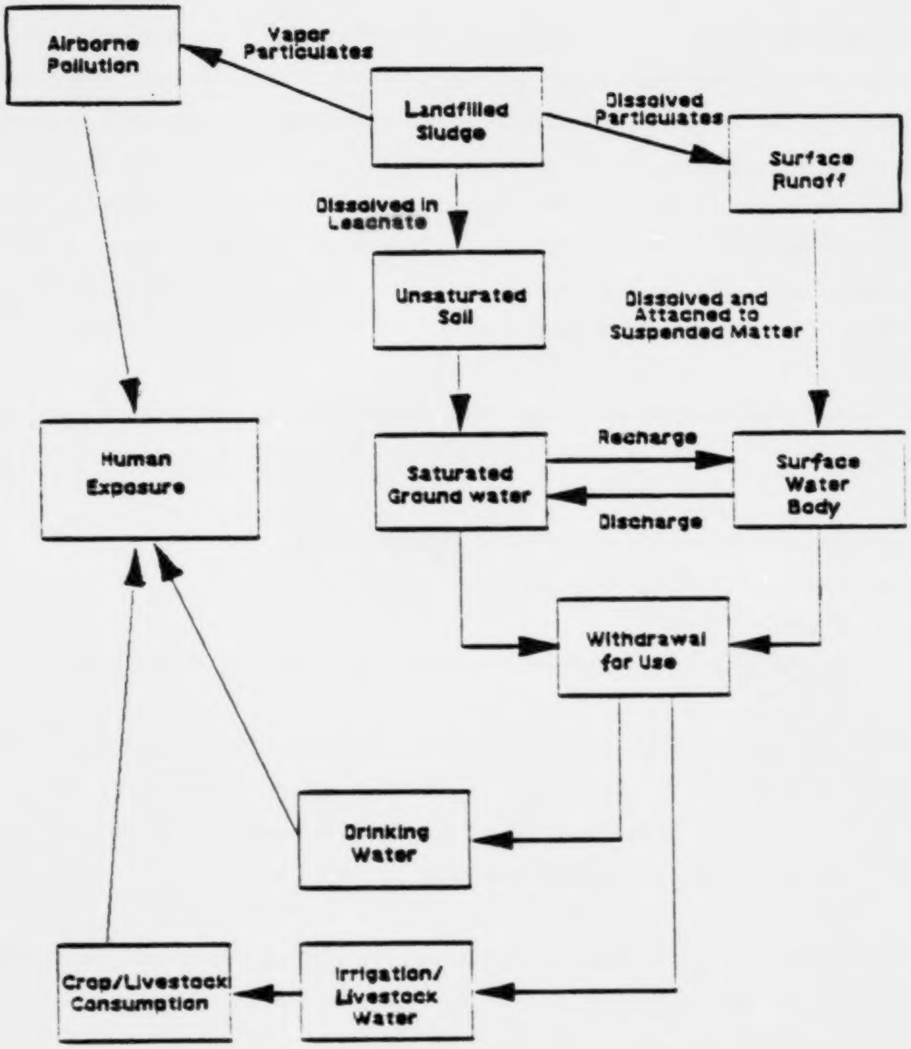
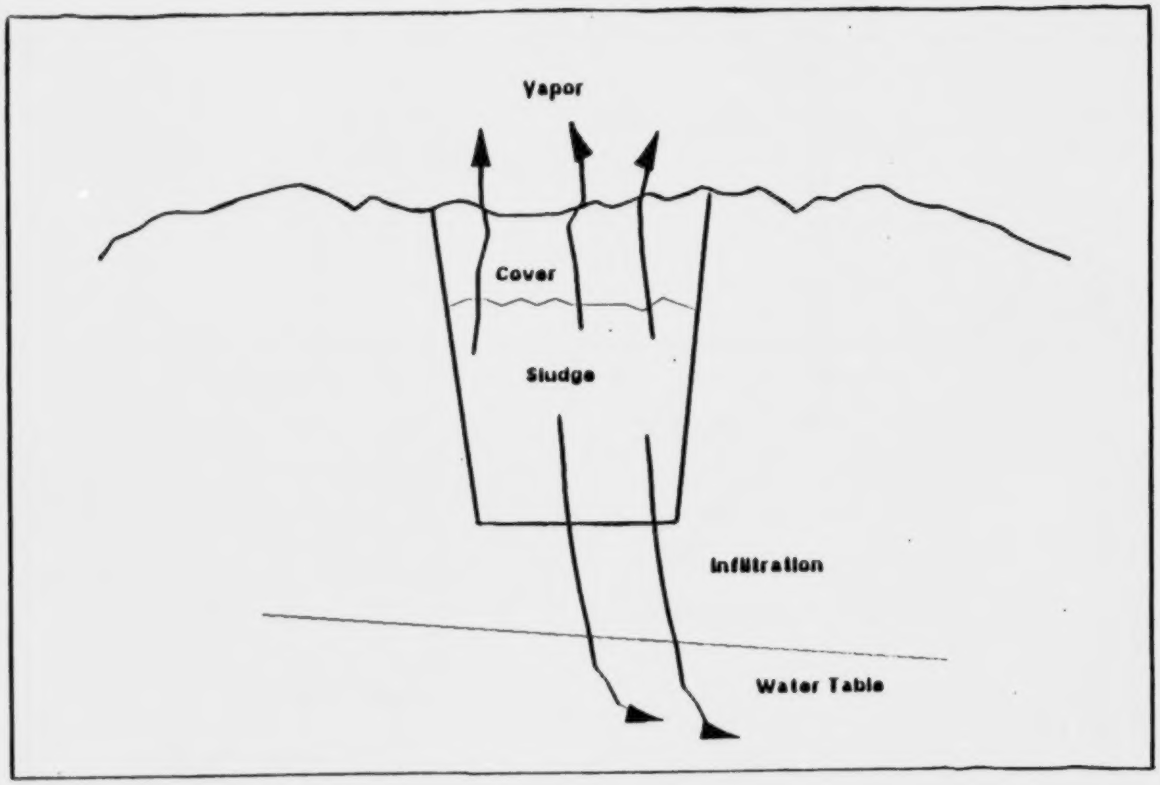

Airborne Pollution
Vapor Particulates
Landfilled Sludge
Dissolved Particulates
Surface Runoff
Dissolved in Leachate
Unsaturated Soil
Dissolved and Attached to Suspended Matter
Human Exposure
Saturated Ground water
Recharge
Discharge
Surface Water Body
Withdrawal for Use
Drinking Water
Crop/Livestock Consumption
Irrigation/ Livestock Water

FIGURE 2-1. Routes to Human Exposure from Landfilling Sludge



Vapor
Cover
Sludge
Infiltration
Water Table

FIGURE 2-2. Contaminant Migration Pathways for Narrow Trench Landfills

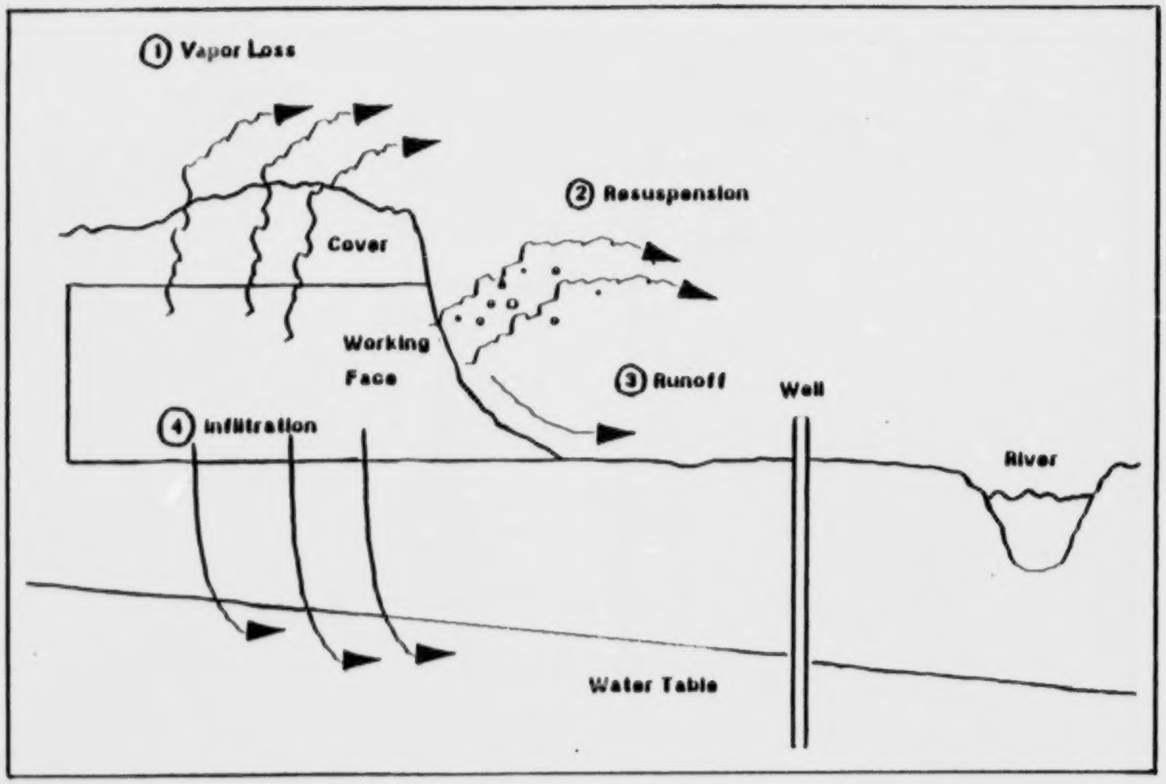

① Vapor Loss
② Resuspension
Cover
Working Face
③ Runoff
Well
④ Infiltration
River
Water Table

FIGURE 2-3. Contaminant Migration Pathways for Pit or Wide Trench Landfillls

)

Vapor loss of volatile compounds is also a very likely source of contamination. Vapor loss may be caused by volatilization from the uncovered working face of the landfill or contaminant releases from within the fill and subsequent migration of these contaminants through the soil cover. Vapor loss from sewage sludge landfills was identified as a possible source of contamination from benzene, cyanide, dimethylnitrosamine, and trichloroethylene. The MEI for the vapor pathway was defined as an adult residing at the property boundary of the landfill who inhales vapors from the site 24 hours (hr) a day for his or her entire 70-yr lifespan.

The third pathway, surface runoff, is one of the least likely routes of exposure because:

- Soils are used for cover, making the working face the only significant source area for contaminated runoff.
- At landfills where the working face is below the grade of the surrounding areas, runoff will be contained by the facility design.
- Where the working face is not below grade, the regulation requires that drainage be contained in the downflow direction (runon controls).

The fourth pathway, suspension of contaminated particles from the working face, is also an unlikely route of exposure because the face of the fill will not be exposed for more than 8-12 hr/day and because threshold wind speeds necessary for suspension rarely would be exceeded for any length of time. This pathway, as well as the surface runoff pathway, is not modeled by SLUDGEMAN.

2.3 OVERVIEW OF SLUDGEMAN

SLUDGEMAN incorporates a number of computer models and submodels that are designed to represent the leaching of contaminants from a source and the fate and transport of the leached material through a zone of unsaturated soil (where the spaces between soil particles are filled with both air and water) and into and through a saturated zone, typically considered an aquifer. The model also predicts the release of volatile contaminants into the atmosphere. SLUDGEMAN's major components are:

- Two models that determine the rate at which each contaminant of concern leaches from the monofill (SATM1 and SLUGMAIN)
- A model that determines the rate of contaminant transport through the unsaturated zone and the processes that occur in this zone to degrade contaminants or retard transport (CHAIN)
- A model that predicts the transport and fate of contaminants in the saturated zone (AT123D)
- A model that integrates CHAIN and AT123D (AT123DIN)
- Two models (SATIT and ALLCON) that (1) calculate the rate of release to and the concentrations of volatile contaminants in the atmosphere and (2) determine the allowable concentration of contaminants in sludge using the results of both the groundwater and vapor pathway modeling

Section 2.4 discusses the methodologies incorporated in the five major groundwater models (SATM1, SLUGMAIN, CHAIN, AT123DIN, and AT123D), Section 2.5 discusses the two vapor release models, and Section 2.6 presents the methodology used to back-calculate maximum concentrations of contaminants in sludge based on the criteria and the predictions of the SLUDGEMAN models. Finally, Section 2.7 presents a summary of the comments generated by SAB's review of SLUDGEMAN and EPA's responses to those comments.

SLUDGEMAN performs two types of analyses using the component models listed above. For monofills located over Class I aquifers, SLUDGEMAN uses SATM1, CHAIN, and SATIT to predict groundwater concentrations and vapor loss and to back-calculate allowable concentrations of contaminants in sludge. The Class I aquifer analysis assumes the leachate is not transported in the aquifer, but is ingested by the MEI in the same concentration at which it enters the aquifer. Thus AT123D, which models transport through the saturated zone, is not used. Class I aquifers are treated very conservatively because they are highly vulnerable to contamination, being either irreplaceable as sources of drinking water to substantial populations or ecologically vital.

For monofills located over Class II aquifers (ground water that is not classified as Class I, and that either is used currently or is available as a potential source of ground water), the model uses SLUGMAIN instead of SATM1 to calculate leach times. The model for monofills over Class II aquifers also uses the saturated zone transport model AT123D and the unsaturated transport model CHAIN. Additionally, the ALLCON model, rather than SATIT, is used to predict volatile contaminant concentrations in the atmosphere and to calculate allowable sludge concentrations. During transport in the saturated zone (aquifer), contaminants can be diluted or degraded; thus the criteria generated are less stringent than those generated using the Class I analysis.

A few additional analytical distinctions are made in modeling the two types of monofills. For monofills located over Class II aquifers, SLUDGEMAN calculates downgradient pollutant concentrations on the centerline of the monofill rather than on the edge of the monofill, because the highest pollutant concentrations are expected along the centerline. The model also assumes the monofill is contiguous to one edge of the aquifer, so that pollutants are modeled to disperse laterally into the center of the aquifer from the edge. This approach yields higher downgradient concentrations than those calculated assuming the monofill were centered in the aquifer with lateral dispersion occurring in both directions toward the sides. The assumption that the monofill is located contiguously to one edge of the aquifer is realistic because monofills can be placed in any location relative to an aquifer, including near one edge.

Monofills can be located over other types of aquifers, including Class IIIA and IIIB aquifers and their subclassifications. SLUDGEMAN does not distinguish monofills located over Class IIIA and IIIB aquifers from those over Class II aquifers. Class III aquifers are not used as sources of drinking water; Class IIIA aquifers yield insufficient water or have solids content greater than 10,000 milligrams (mg/L), and Class IIIB aquifers are contaminated naturally or by human activity and cannot be cleaned up using reasonable water treatment methods.

2.4 DESCRIPTION OF THE GROUNDWATER COMPONENTS OF THE SLUDGEMAN MODEL

SLUDGEMAN's groundwater components consist of a number of sequential models that vary depending on the type of aquifer to be modeled. These models include leachate pulse-time calculations (SATM1 and SLUGMAIN), an unsaturated zone fate and transport model (CHAIN), a saturated zone fate and transport model (AT123D), and a model that integrates CHAIN and AT123D (AT123DIN). A discussion of each of these models follows.

2.4.1 Pulse-time Calculation (SATM1 and SLUGMAIN)

Leaching of contaminants out of a monofill does not happen instantly. The leachate pulse time is the length of time from the beginning to the end of leaching, i.e., when all leachable materials have migrated from the monofill. The SATM1 model is used to calculate the pulse time in Class I aquifer simulations; SLUGMAIN is used to calculate pulse time in Class II aquifer simulations. The input data required for these calculations include: concentrations of pollutants in the leachate, concentrations of pollutants in the sludge (dry weight), sludge moisture content, sludge storage capacity, sludge density, net recharge rate, width and length of the landfill, and height of the fill material. These parameters and assigned values are discussed in detail in Section 3.

Several assumptions are made in calculating pulse time: (1) the estimated leachate concentrations are an approximation of leachate strength, (2) all the contaminants ultimately solubilize, and (3) the leachate pattern is a pulse of equal height throughout its duration (i.e., a square wave, rather than a curve representing a gradual increase followed by a gradual decline in the amount of contaminants leaching from the monofill). The results of the pulse-time calculation are used as an input to CHAIN, which calculates contaminant concentrations in the unsaturated zone. The parameters total contaminant levels in the sludge, contaminant concentrations in the leachate, sludge moisture content, and recharge rate are used to calculate pulse time. These factors are relevant to pulse time according to:

$$Q = M/X \qquad (1)$$

and

$$Q = R\hat{T} + (L - S) \qquad (2)$$

where Q = volumetric water flow of leachate per square meter (m^2) required for the contaminant to be completely leached; the units for Q are in m^3

R = recharge of infiltrate volume entering the landfill, in m^2/yr. This can be calculated as: R = P - ET - RO, where P is precipitation, ET is evapotranspiration loss, and RO is runoff

S = storage capacity for water in sludge, defined as the "dry" water content per m^2 for the sludge under normal atmospheric conditions, i.e., the product of fill height after drainage, sludge density, and moisture content divided by 1,000 kilograms (kg)/m^3

L = water content of sludge per m^2 at time of disposal, i.e., the product of fill height, sludge density, and moisture content divided by 1,000 kg/m^3; the units for L are in m^3

$\hat{T}$ = the pulse time over which all contaminant will be released from the sludge, in yr

M = mass of contaminant contained in a volume of sludge represented by the height of the sludge in the fill and a m^2 cross section, i.e., M = (height of fill x 1.0 m^3) x (density of sludge, kg/m^3) x (concentration of contaminant in sludge; kg/kg) x (1 - moisture content); the units for M are in kg

X = concentration of leachate contaminant in kg/m^3

All of these parameters are discussed more fully in Section 3.

When equations 1 and 2 are combined, the pulse time $\hat{T}$ can be calculated:

$$\hat{T} = \frac{M - X(1 - S)}{XR} \qquad (3)$$

For degradable contaminants, the initial mass of contaminant, M, changes with time. If a first-order decay mechanism is assumed at a degradation rate, d, the equation becomes:

$$T = \frac{-1}{d} [\ln \frac{XR}{XR - dM}] \qquad (4)$$

where T - pulse time, yr
d - degradation rate constant, yr^{-1}
X - leachate concentration, kg/m^3
R - recharge rate, m^3/yr
M - mass of contaminant in sludge, kg

2.4.2 Unsaturated Zone Model (CHAIN)

CHAIN is the analytical model used to solve the one-dimensional convective-dispersive transport equation for the unsaturated zone (Van Genutchen, 1985). This model predicts the way dispersion elongates the leachate pulse as it moves through the unsaturated zone, resulting in decreasing contaminant concentrations. The CHAIN model itself is composed of several submodels. These submodels calculate the length of time it takes the leachate to travel through the unsaturated zone to the saturated zone below, and the rates of retardation and degradation of contaminants in the unsaturated zone. Length of travel time, retardation, and degradation calculations used in the CHAIN model are discussed below.

The MINTEQ model, which predicts the fate of dissolved metals, is also discussed in this section. MINTEQ, which is not part of CHAIN, was run separately, and the results were used to adjust the metals concentrations predicted by CHAIN to account for precipitation of metals from the leachate. These adjustments are "hard-wired" into the CHAIN model and do not need to be input.

2.4.2.1 Travel-time Calculation

The unsaturated zone consists of the layers of soil between the landfill and the uppermost aquifer. To determine the concentration of a contaminant at the base of the unsaturated zone, the travel time must be calculated. The travel time is the length of time it takes the leachate to travel through the unsaturated zone; it is determined using the depth to ground water and the net recharge rate.

Two basic approaches can be used to determine travel time in the unsaturated zone: (1) analytical models and (2) unsaturated flow models. Both approaches are based on the same fundamental equations but differ in the simplifying assumptions made to solve the equations. Analytical models require fewer data, are computationally easier to use, and generate solutions much more rapidly than unsaturated flow models. Consequently, the Agency selected an analytical model to predict unsaturated zone concentrations of contaminants. Analytical models, however, require the assumption of steady-state conditions. As a result, they predict a constant contribution of leachate over time rather than the periodic storm event contributions that actually occur. This simplifying assumption generally leads to results that overpredict velocity, underpredict travel time and degradation, and overpredict concentrations in the unsaturated zone.

Analytical solutions of travel time through the unsaturated zone are based on Darcy's law for one-dimensional flow. Darcy's law is a mathematical

$$V_z = K(\psi_m) \frac{\delta\psi}{\delta z} \qquad (5)$$

where V_z = seepage velocity in the vertical direction
ψ_m = matric potential (suction head or negative pressure head)
$K(\psi_m)$ = hydraulic conductivity as a function of matric potential
$\delta\psi/\delta z$ = hydraulic gradient in the vertical direction

In unsaturated flow, both hydraulic conductivity and moisture content are nonlinear functions of pressure head. Hydraulic conductivity, moisture content, and pressure head need not be constant throughout a soil column; however, if they are not, a direct analytical solution of Darcy's equation is not possible for unsaturated flow. To solve Darcy's equation for travel time in the unsaturated zone, the following assumptions must be made:

- There is one-dimensional flow in the vertical direction.
- Water flow is steady state.
- Water table conditions exist at the lower boundary.
- The upper boundary has a constant flux.
- Soil characteristics (moisture content versus matric potential and hydraulic conductivity versus matric potential) are constant with depth.
- The hydraulic gradient is sloped vertically downward and equals unity (drainage is caused strictly by gravity, or $\delta\psi/\delta z = 0$).

For nonhomogeneous soils, the constant property assumption can be approximated by dividing the soil profile into a series of layers, then calculating travel time for each layer individually.

The unit gradient assumption greatly simplifies the analysis. Under this assumption matric potential and, therefore, moisture content and hydraulic conductivity, remain constant with depth. When this assumption is made, moisture content can be solved directly in terms of flux through the system and saturated soil properties. When the moisture content and flux are known, the pore water velocity and the travel time through the unsaturated zone can be calculated. The unit gradient assumption generally is valid if gravitational forces dominate other forces (e.g., capillary forces).

If the unit assumption is not made, the analytical solution to unsaturated flow becomes more complex. In this case, calculations for pressure head and moisture content must be solved iteratively. Deriving iterative solutions is a time-consuming task, but can be simplified by using a computer.

All analytical solutions for travel time through the unsaturated zone are one-dimensional. When applying these solutions to specific sites, the horizontal variability of soil characteristics must be considered. If soil characteristics vary spatially, the solution should be applied to the soil profile having the highest hydraulic conductivity. The solution will then yield the highest velocity and shortest travel time (i.e., worst case) for the unsaturated flow system. The solution, which is the water travel time (TW), must then be modified by a retardation factor. A more detailed treatment of the theory and equations used in travel time calculations may be found in the Technical Guidance Manual for Calculating Time of Travel (TOT) in the Unsaturated Zone (EPA, 1985c).

2.4.2.2 Retardation Factor

Contaminants will travel either with the leachate or at a slower velocity depending on the degree to which they are adsorbed onto soil particles. Retardation is a measure of how slowly a contaminant moves through the unsaturated zone with respect to the bulk of the leachate. The retardation factor is calculated from model input data on the bulk density of the unsaturated zone material, the partition coefficient of the contaminant, hydraulic conductivity, effective porosity, effective bulk density, slope of the curve plotting matric potential versus moisture content, and the saturated moisture content of the unsaturated zone material. These input parameters and their assigned values are discussed in Section 3.

The retardation factor (R) for a particular contaminant can be calculated by the equation:

$$R = 1 + (B/\theta)(K_d) \qquad (6)$$

where B/θ = soil-to-solution ratio (bulk density of the soil divided by its effective porosity)

K_d = distribution coefficient

K_d can be measured in the laboratory or obtained from the literature for a wide range of soil types and contaminants. Values for K_d are discussed in Section 3.

2.4.2.3 Degradation

The travel time for a contaminant through the unsaturated zone (TC) can be estimated as the water travel time (TW), derived in the travel time calculation, times the retardation factor for that particular contaminant, as represented by:

$$TC = TW \times R \qquad (7)$$

Reductions in pollutant concentration due to degradation processes, such as hydrolysis and biochemical oxidation are characterized by a degradation constant, d. This constant can be related to the environmental half-life of a pollutant, represented by $t_{1/2}$, the time required for the contaminant concentration to be reduced to one-half its initial value. If a first-order decay mechanism is assumed, the concentration X at any time can be defined as:

$$X = X_o e^{-dt} \qquad (8)$$

where X_o - the initial concentration of X
t - time

Therefore, the half-life, $t_{1/2}$, can be derived as:

$$\ln \frac{X_o}{X} = d_{t_{1/2}} \qquad (9)$$

or

$$t_{1/2} = \frac{\ln 2}{d} \qquad (10)$$

Equation 8 can be used to determine the degree of degradation that will occur as the leachate moves through the unsaturated zone. In this way, the leachate concentration from the source, X, can be converted to the value predicted at the point where the leachate enters the aquifer. This conversion is accomplished by inserting the contaminant travel time in the unsaturated zone, TC, into Equations 7 and 8:

$$X = \bar{X} e^{-dTC} \qquad (11)$$

or

$$X = X e^{\frac{(-\ln 2TC)}{t_{1/2}}}$$

The resulting value for X is the contaminant concentration that should be applied to all subsequent saturated zone transport calculations. The above equation gives the pollutant concentration reduction in the unsaturated zone caused by degradation. That is, the calculation conservatively assumes that the contaminant plume moves as a pulse through the unsaturated zone, with no dispersion. In actuality, dispersion elongates the contaminant pulse as it moves through the unsaturated zone, with a resulting decrease in concentration.

2.4.2.4 Metals Modeling (MINTEQ)

After CHAIN is used to determine contaminant concentrations at a depth equal to the depth to ground water for a period of several contaminant travel times, the concentrations of metals must be adjusted to account for any precipitation of metal compounds that exceed solubility limits. The MINTEQ model is used to calculate the mass distribution of a dissolved metal between various uncomplexed and complexed aqueous species and to predict the precipitation and dissolution of these species (Felmy et al., 1984). MINTEQ was run for a wide range of conditions, and the results were programmed into the Agency's landfill model.

When the pH and Eh (oxidation-reduction potential) of the ground water are known, CHAIN can automatically adjust metal concentrations at the base of the unsaturated zone using the MINTEQ calculations for those conditions. After these adjustments are made, CHAIN compares the maximum concentrations of metals and organic contaminants to the drinking water criteria. For Class II groundwater analyses, if the maximum concentrations are below the criteria, the modeling analysis can be concluded without analyzing saturated zone transport. If the predicted concentrations exceed the criteria, then contaminant transport in the ground water must be simulated.

2.4.3 Saturated Zone Model (AT123D)

Analytical solutions and numerical modeling are the two basic approaches for estimating contaminant travel time and concentrations in saturated groundwater flow systems. The Agency selected an analytical groundwater model, AT123D, because this approach requires fewer data, is less time consuming to establish and run, and requires no expensive equipment or specialized expertise.

AT123D is the analytical model used to solve the advective-dispersive transport equation for the saturated zone (Yeh, 1981). Advection accounts for the movement of solutes with the mean velocity of the water. Dispersion of the leachate plume occurs because solutes travel more slowly near the walls of the soil pores than in their centers and faster in larger pores. Molecular diffusion also is included in AT123D to account for spreading of the leachate plume due to random molecular motion. Additionally, AT123D calculates the sorption and decay of contaminants. Input data required for the model include the type of material in the saturated zone, mixing thickness of the aquifer, hydraulic conductivity, effective porosity, hydraulic gradient, effective bulk density, width of the saturated zone, partition coefficients, decay rates, background groundwater concentrations for each contaminant, and the distance to the compliance point. The algorithms incorporated in this model are not presented here. The interested reader can find a discussion of the highly complex mathematical expressions used to model saturated transport modeling in Yeh (1981). The parameters used as input to the model and their assigned values are discussed in Section 3.

The Agency's goal is to field-validate all the risk-based models used to support the proposed rule. AT123D has not yet been field-validated, but the theory behind the model is well known and accepted by groundwater experts, and it is widely used by the Agency to support regulations and investigations by EPA's Office of Toxic Substances and Office of Solid Waste.

2.4.4 Integrating the Unsaturated and Saturated Zone Models (AT123DIN)

To integrate between the CHAIN unsaturated zone model and the AT123D saturated zone model, the contaminant concentrations output from CHAIN must be converted to mass flux rates from an appropriate repository. The repository is defined as the area below the landfill to a depth in the aquifer equal to the mixing depth of the leachate from the landfill. The amount of contaminant mass leaching into the aquifer annually is the concentration of each contaminant times the recharge rate times the area of the landfill. The annual mass flux exiting the area below the landfill is either the total annual mass flux, if the groundwater flow rate is sufficient to carry the total mass of leachate away each year, or a fraction of this amount equal to the fraction of the landfill length that the ground water travels in a year. The mass flux from the area below the landfill is therefore equal to the contaminant concentration times the recharge rate times the landfill width times the minimum of the landfill length and the groundwater velocity. The result of this calculation is the mass flux value needed as input to AT123D.

If the groundwater flow is insufficient to carry away all the contaminant leachate that enters the aquifer each year, then the release time for contaminants into the aquifer is increased by the ratio of the length of the landfill to the annual groundwater travel distance. This adjustment accounts for the increased length of time required to wash away all contaminants from beneath the landfill.

2.5 DESCRIPTION OF THE VAPOR PATHWAY COMPONENTS OF THE SLUDGEMAN MODEL

Vapor loss from landfills has been identified as a potential problem for certain volatile toxic chemicals such as benzene, cyanide, dimethylnitrosamine, and trichloroethylene. To estimate landfill vapor loss, the Agency is using (in SATIT and ALLCON) the same analytical model that was developed to evaluate vapor loss and dispersion of contaminants from hazardous waste sites in the landfill ban analysis (Environmental Science and Engineering, 1985). The vapor loss models consider three periods: (1) the operating period with uncovered wastes, (2) the period of shallow temporary cover, and (3) the post-closure period with permanent cover. The models assume that pollutants will evaporate from the monofill for a total of 70 yr. The monofill is assumed to be active for 20 of those 70 yr, during which the sludge deposited each day is assumed to remain open to the air for an average of 4 hr. The sludge is therefore assumed to be open to the air 1/6th of the time for the first 20 years, covered with a temporary cover for 5/6ths of that time, and covered with a final cover for 50 yr. Degradation and deposition are not considered because travel times are relatively short and the Agency wanted to be conservative in estimating potential vapor concentrations.

For the operating period, during which wastes remain uncovered, the model assumes that volatilization depends directly on wind speed. The model predicts the maximum possible exposure to vapor concentrations because it assumes that (1) wind speed and direction are constant, (2) stable atmospheric conditions prevail, and (3) the receptor is located downwind and along the centerline of the plume.

For the two periods when wastes are covered, the model assumes that the loss rate of contaminants is independent of wind speed and is controlled by diffusion of contaminants through the soil. The model assumes that the final soil cover applied to a landfill cell has the same permeability as that of the temporary soil cover. In practice, because the final soil cover is usually less permeable, the model assumption leads to an overprediction of loss rates.

The rate at which pollutants evaporate from a monofill is estimated using Henry's law constants, a method that also leads to overprediction. Henry's law states that the mass of any gas that dissolves in a given volume of liquid at constant temperature is directly proportional to the pressure that the gas exerts above the liquid. The use of Henry's law can be avoided if adequate data specifying the vapor pressure of each contaminant are available. Because vapor pressures are not measured routinely, the model uses Henry's law to specify vapor concentration as a function of liquid concentration. Henry's law is most appropriate for low concentrations of dissolved pollutants and low solids content sludges. As pollutant concentrations and solids content increase, Henry's law tends to overpredict vapor pressure as a result of activity effects and partitioning between solid and liquid phases.

The models are written to calculate vapor concentrations for six of the ten organics of concern: benzo(a)pyrene, bis(2-ethylhexyl)phthalate, chlordane, DDT/DDD/DDE, dimethylnitrosamine, and polychlorinated biphenyls

(PCBs). Dimensionless Henry's law constants and molecular weights have been incorporated into the model for these chemicals.

In addition to Henry's law constants and the molecular weight of contaminants, other input data are required for the vapor loss models, including average wind speed and air temperature, porosity of cover soil, cover thickness, length or width of source area, and distance from the center of the source to the compliance point. These parameters and the values assigned to them are discussed in Section 3.

2.6 USE OF SLUDGEMAN TO ESTABLISH CRITERIA FOR POLLUTANTS IN SLUDGES

The models in the risk assessment methodology previously detailed can incorporate data from any site to determine if the resulting pollutant concentrations at a specific point of compliance exceed the human health criteria. The Clean Water Act (CWA), however, requires EPA to establish numeric criteria for the pollutants in the sludge. The model must therefore be able to calculate in the reverse mode as well, i.e., starting with the human health criterion and working backwards to determine the sludge pollutant concentrations that will not exceed the human health criterion at the point of compliance. Both the groundwater and vapor pathways must be considered in these calculations. ALLCON and SATIT, after calculating vapor loss, use the predictions from the groundwater models combined with the vapor loss predictions to calculate the total maximum allowable concentrations of contaminants in monofilled sludge.

2.6.1 Groundwater Pathway Calculation

For a given release time, with no sorption at the unsaturated/saturated zone interface due to Eh/pH differences, the relationship between leachate concentration from a monofill and concentration at the downgradient monitoring point is linear: a given change in leachate concentration produces a corresponding change in outflow concentration. Thus, for organics, given the acceptable limits for outflow concentration, the allowable leachate concentration can be calculated. The CWA, however, specifies that numeric criteria are to be based on concentrations in sludge, not in leachate. Therefore, the limiting leachate concentration must be related back to a limiting sludge concentration.

For organic contaminants, the relationship between their concentrations in sludge and their concentrations in leachate is defined by the distribution coefficient K_d. This coefficient defines a constant relationship between the sludge concentration (S_c) and the leachate concentration (L_c) such that $S_c = L_c \times K_d$. The partition coefficient K_d and its derivation are discussed in Section 3. With K_d and the limiting leachate concentration (L_d) determined by the model, the limiting sludge concentration (S_d) is calculated from $S_d = K_d * L_d$.

For metal constituents, the relationship between contaminant concentration in sludge and that in leachate is more difficult to define. Metal concentrations in leachate are often defined by solubility constraints rather than by partitioning. In this case, leachate levels will stay at the solubility threshold regardless of the sludge concentration. It is the duration of the release of metals at the solubility limit that affects the outflow concentration at a monitoring point. Given a constant recharge rate, chemical release time is governed by sludge pollutant concentration. The higher the sludge pollutant concentration, the longer the chemical is released, and the more the outflow concentration approaches the leachate concentration or solubility limit. The model conventionally assumes that the highest reported effluent or leachate value for the metal constituent defines the solubility limit. The sludge concentration can then be varied to determine the inventory of contaminants producing maximum acceptable dose levels at the point of exposure.

Different types of sludges will affect a metal's chemistry differently and, therefore, its maximum solubility level in leachate. To accommodate this sludge-specific factor, a leachate test can be conducted to determine empirically the actual leachate concentration. Then the relationship derived from the sludge's total contaminant level and the leachate concentration can be used to convert the limiting leachate concentrations to a sludge criteria for that site.

For some metals, the back-calculation is not straightforward due to complicated geochemistry. Also, the sorption characteristics in the aquifer are not linear. The relationship between the dry-weight concentration of a pollutant in the sludge and the final concentration at the point of compliance is therefore nonlinear.

Because of this nonlinear relationship, an iterative approach is required to solve for the allowable dry-weight concentrations in sludge. For the iterative process, an initial set of typical dry-weight concentrations are input into the model. Using these inputs, the model then calculates output concentrations at the point of compliance, which are compared to the human health criteria for each pollutant. The dry-weight concentrations for each pollutant are then adjusted appropriately, and ALLCON writes the new dry-weight concentrations to a new input file to be used in another model run. The model is rerun, the new results are compared to the human health criteria, the dry-weight concentrations are adjusted again, and a new input file is written. After several such model runs, the output concentrations of contaminants should approach the human health criteria and the dry-weight concentrations of contaminants should approach the allowable sludge dry-weight concentrations.

If no upper limit to the dry-weight concentration for a given pollutant appears to be generated, the model limits the dry-weight concentration to 10,000 mg/kg (1% pollutant by weight, a reasonable upper limit). This step prevents the model from spending needless time calculating unreasonable dry-weight concentrations of contaminants in cases, for example, in which the leachate concentration of a pollutant is less than its human health criterion.

2.6.2 Vapor Pathway Calculation

Because total pollutant concentrations at the point of compliance are required, the calculated vapor concentration must be added to the calculated leachate and background concentrations of the groundwater pathway for each pollutant. The model calculates vapor concentrations in mg/m^3, whereas liquid concentrations are calculated in mg/L. Therefore, the vapor concentrations must be converted to equivalent liquid concentrations before they can be summed to produce a total concentration for comparison with the human health criterion.

The model assumes that a 70-kg person breathes about 20 m^3 of air per day and drinks about 2 L of water per day. Thus, if the air contains 1 micrograms $(\mu g)/m^3$ of a pollutant, the person is breathing 20 μg of the pollutant per day. Because the person is assumed to drink 2 L of water per day, inhaling 20 μg of a contaminant daily is equivalent to drinking water containing 10 $\mu g/L$ of that contaminant. Ten $\mu g/L$ is equivalent to 0.01 mg/L. Thus, 1 $\mu g/m^3$ vapor concentration is equivalent to 0.01 mg/L liquid concentration. Under these assumptions, the model multiplies the vapor concentration in $\mu g/m^3$ by 0.01 to covert to equivalent liquid concentrations in mg/L. The pollutant concentrations then can be summed for comparison with the human health criterion.

2.7 SCIENCE ADVISORY BOARD REVIEW

The SAB reviewed the groundwater and vapor modeling used in SLUDGEMAN. The SAB had four major recommendations regarding the groundwater modeling:

- The unsaturated zone should be modeled with PRZM (another unsaturated zone model) instead of CHAIN.
- The unsaturated zone should be modeled assuming anaerobic conditions.
- The model should be modified to simulate the effect of liners.
- The model should be expanded to include the groundwater-to-surface water pathway and implications for the wildlife food chain, crop uptake, and bioaccumulation by edible aquatic organisms.

The CHAIN model predicts constant infiltration over time. This steady-state approach tends to overpredict velocity, underpredict travel time and degradation, and overpredict contaminant concentration. PRZM uses a curve-number approach derived by the U.S. Department of Agriculture's Soil Conservation Service to distribute daily rainfall into runoff and infiltration. Infiltrating water cascades downward to successively deeper layers as the soil water content of each layer exceeds field capacity. PRZM's dynamic approach allows consideration of pulse loads and prediction of peak events and estimates time-varying mass emission or concentration profiles.

The Agency's steady-state approach was designed to predict long-term effects over the MEI's 70-yr lifespan. Short-term fluctuations in pollutant concentrations predicted by PRZM average out in such an analysis. The steady-state CHAIN model is adequate for the slowly degrading chemicals currently being regulated for sludge disposal in landfills. In the future, if more rapidly degrading chemicals are added to the regulation, the effects of short-term fluctuations will be evaluated using a dynamic modeling approach such as PRZM.

The Agency agreed with the SAB recommendation that anaerobic conditions should be assumed for the unsaturated zone below a sewage sludge landfill. The dissolved organics that leach out of these landfills are expected to produce an anaerobic environment. The current version of the Agency's landfill model includes anaerobic biodegradation rates and hydrolysis rates for the regulated pollutants.

The SAB also suggested that the Agency's landfill model simulate the effect of liners on leachate quality and quantity. The Agency decided against this suggestion because liners are not being required in this regulation and generally are not in use at sewage sludge monofills. The model is based on the conservative assumption that no liner is present, so that the calculations will yield the most protective maximum allowable sludge concentrations.

The Agency decided to postpone action on the SAB recommendation that the model be expanded to consider the groundwater-to-surface-water pathway. The Agency is convinced that the existing groundwater and vapor analysis accounts for the most likely pathways of human contamination. The Agency is requesting public comment on the need to consider the groundwater-to-surface-water link and its effects on human and ecosystem health. Based on public comments, the Agency will consider adding this surface-water pathway to the model.

SECTION THREE

DATABASE FOR THE RISK MODEL

Because no risk assessment methodology or risk model has been developed for surface disposal sites, there are no known factors for which to develop a database. The surface disposal site model, however, should require a database similar to that developed for the monofilling disposal option. The surface disposal model, like SLUDGEMAN, will be used to calculate the pollutant concentrations in sludge that will allow a particular risk level (specified as a set of human health or environmental impact criteria) to be met.

Before the new model could be run, however, a number of parameters would have to be specified. These parameters will probably be similar to those for monofill; for SLUDGEMAN the parameters defined (1) a model sludge type; (2) groundwater conditions, saturated and unsaturated zone conditions, and landfill surface dimensions for the hypothetical site; and (3) chemical-specific factors, such as sludge contaminant concentrations. This section presents the values assigned to each parameter in modeling monofills and discusses the basis for the choice of each value. Table 3-1 presents a summary of all the parameters to be defined with their assigned values (except for the chemical-specific parameters, which are too numerous to list here). Some of these values are input by the user; others are "hard-wired" into the program.

3.1 MODEL SLUDGE PARAMETERS

The model requires that a specific sludge type be defined. Sludges are defined primarily by their water or solids content, their storage capacity,

TABLE 3-1. Summary of Model Parameters and Assigned Values

Parameter	Value
Sludge Parameters	
Sludge moisture content	0.8 kg/kg
Sludge storage capacity	0.90 kg/kg
Sludge density	1,025 kg/m^3
Specific gravity (solids)	1.125
General Hydrologic Parameters	
Net recharge	0.5 m/yr
Depth to ground water (Class I; other)	0 m; 1 m
Aquifer thickness	15 m
Ground water pH	6 su
Ground water Eh	500 mv
Saturated soil hydraulic conductivity	10,000 mg/yr
Unsaturated Zone Parameters	
Unsaturated soil type	Sand
Bulk density in saturated zone	2,380 kg/m^3
Effective porosity in saturated zone	0.1 m^3/m^3
Saturated zone hydraulic conductivity	2,000 m/yr
Hydraulic gradient in saturated zone	0.005 m/m
Surface Parameters	
Landfill site geometry	100 m x 100 m x 3.46 m Cell length = 8 m
Distance to property boundary	150 m
Surface wind velocity	1 m/sec
Air temperature	15° C
Air-filled porosity of cover soil	0.1 m^3/m^3
Total porosity of cover soil	0.4 m^3/m^3
Cover thickness (active and final)	0.3 m; 1 m
Chemical-specific Parameters	
Concentrations of sludge contaminants in sludge	See Sec. 3.6.1
Concentrations of contaminants in sludge leachate	See Sec. 3.6.2
Distribution coefficient	See Sec. 3.6.3
Saturated and unsaturated zone decay rate	See Sec. 3.6.4
Background concentrations of contaminants	See Sec. 3.6.5
Health effects levels	See Sec. 3.6.6
Molecular weight	See Sec. 3.6.7
Henry's law constants	See Sec. 3.6.8

their density, and their specific gravity. The parameters that define the model sludge follow.

3.1.1 Sludge Moisture Content

The value selected to represent the model sludge moisture content is 0.8 kilograms (kg)/kg, which indicates a sludge with a solids content of 20%. This value was selected because a solids content of 20% is required for sludges codisposed in municipal landfills and because most types of sludge landfills commonly take sludges with solids contents of about 20%. Table 3-2 presents typical solids contents of sludges disposed in a number of different types of monofills.

3.1.2 Sludge Storage Capacity

The storage capacity for water in sludge is the moisture content of the sludge when it has been allowed to drain completely. It is defined as the "dry" water content per square meter (m^2) of the sludge under normal atmospheric conditions.

For the risk model, the storage capacity will be set equal to the typical moisture content of gravity-thickened sludge, that is, 0.90 kg/kg. The selection of this value is conservative because it represents a high storage capacity and, thus, a moister sludge. In moister sludges, more water is available to move through the overburden into the ground water, and pollutants are more likely to reach ground water faster. Thus, the health and environmental risks associated with sludges having a high storage capacity are greater than those for sludges with a low storage capacity.

TABLE 3-2. Typical Sludge Solids Content by Landfill Type

Type of Landfill	Percentage of Sludge Solids
Narrow trench	15-28%
Wide trench	20%
Area fill mound	20%
Area fill layer	15%
Diked containment	20%

Source: EPA, 1978.

3.1.3 Sludge Density

Density is defined as the mass of a substance per unit volume. The value selected for sludge density in the model is 1,025 kg/cubic meter (m^3), which represents the sludge's effective density. For sludges, the effective density is a more appropriate measure of density than density as defined above. The effective density takes into consideration the physical state (particle size, amount of bound water, degree of flocculation, etc.) and the chemical composition of the discrete sludge particle as it exists in a sludge mixture.

As the solids content of a sludge increases, the density is even better described as a bulk density. A bulk density calculation considers the entire sludge mixture when figuring density. This type of calculation is not required to determine the density of a sludge with a 20% solids content.

3.1.4 Specific Gravity

The density of the sludge is used to calculate its specific gravity, which is defined as the ratio of the density of a substance to the density of a standard substance. The standard substance used to calculate the specific gravity of the modeled sludge is water, which has a density of 1,000 grams/liter (g/L) under standard conditions of temperature and pressure. Sludge with low solids content has a density similar to that of water, so its specific gravity is close to one. As the sludge solids content increases, the density of the sludge increases because water is displaced with solid materials that have a higher density than water. The density could decrease, however, if the water is displaced by solid material with a relatively lower particle density than water.

The specific gravity of a sludge can be readily determined by simple procedures such as those presented in Standard Methods for the Examination of Water and Wastewater (APHA, 1971). This method involves determining the weight of a given volume of sludge compared to the weight of an equal volume of distilled water. As noted in Standard Methods, free- and nonfree-flowing sludges have different densities; free-flowing sludges have effective densities and nonfree-flowing sludges have bulk densities. These different types of densities must be taken into account when sludge is analyzed.

The specific gravity of sludge can vary widely depending on the state of the sludge (e.g., whether it has been settled, thickened, dewatered, dried, etc.). Therefore, the conditions under which a sludge was sampled should always accompany any statement about its specific gravity, and the analytical characterization methods used to determine the solids content should be specified.

Measuring sludge density is not a simple task; the specific gravity of sludge is usually easier to measure than its density. Rather than measuring density to calculate specific gravity, the models can measure the specific gravity of the sludge solids to derive the specific gravity of the whole sludge, which is the reciprocal of the sludge density. The following equation illustrates this calculation (Eckenfelder and Santhanam, 1981).

$$\frac{(WP)}{SG_{water}} + \frac{(WP)}{SG_{solids}} = \frac{1}{SG_s} \qquad (1)$$

where SG = sludge specific gravity
WP = weight percentage
$1/SG_s$ = sludge density

In the case of the sludge being defined here, which has a 20% solids content, a specific gravity of 1.139 for the sludge solids (chosen to represent a typical value for the modeled sludge) produces the following:

$$\frac{0.8}{1.0} + \frac{0.2}{1.139} = \frac{1}{1.025} \qquad (2)$$

This calculation shows the derivation of the 1.025 kg/m^3 density defined in Section 3.1.3.

3.2 GENERAL HYDROLOGIC PARAMETERS

The model will require a number of inputs that define the general hydrologic conditions of the hypothetical site. These inputs include net recharge, depth to ground water, aquifer thickness, and groundwater pH (relative hydrogen ion concentration) and Eh (oxidation-reduction potential). These parameters are discussed in the following sections, and the values selected for use in the model are presented.

3.2.1 Net Recharge

The primary source of ground water is precipitation, which infiltrates through the ground surface and percolates to the water table. Net recharge is the amount of water per unit of land that penetrates the ground surface and reaches the water table. Consequently, recharge water is available to transport a contaminant vertically to the water table and horizontally within the aquifer. The recharge rate, which is the amount of recharge per year (yr), controls the quantity of water available for dispersion and dilution of contaminants. Recharge is thus a principal vehicle for leaching and transporting solid or liquid contaminants to the water table. The greater the recharge rate, the greater the potential for pollution, up to the point at which the amount of recharge is large enough to dilute the contaminant. At this point, the pollution potential ceases to increase and may actually decrease (EPA, 1985b).

For modeling purposes, the recharge value was set to the highest rate of net infiltration known to occur, 0.5 m/yr, based on information from the Hazardous Waste Management System Land Disposal Restrictions Regulation (51 FR

1602, January 14, 1986). The highest rate of net infiltration was chosen as most conservative because, as discussed previously, pollutants are more likely to reach ground water and travel at a faster rate when the rate of recharge is greater.

3.2.2 Depth to Ground Water

The depth to ground water is generally defined as the depth from the land surface (or the lowest point of the landfill or surface disposal excavation) to the water table. The water table is the subsurface interface between the unsaturated zone (where the pore spaces are filled with water and air) and the saturated zone beneath (where all the pore spaces are filled with water); it may be present in any type of media and may be either permanent or seasonal. Most saturated zones are termed aquifers unless they lack the permeability to yield sufficient water. Only true aquifers are considered when computing depth to ground water.

The depth to ground water is important in evaluating the likelihood that pollutants moving through the unsaturated soil will reach the ground water. It also determines the distance that a contaminant must travel before reaching the aquifer, and it may help to determine the amount of time during which the contaminant maintains contact with the surrounding media. These factors influence the amount of attenuation that may occur as the pollutants are transported. Attenuation processes lessen the amounts or deleterious effects of contaminants. The factors that affect attenuation are the physical and chemical processes and properties that include density, solubility, sorption, biodegradation, oxidation-reduction, dilution, hydrolysis, dispersion, viscosity, mechanical filtration, ion exchange, volatilization, and buffering or neutralization. As depth to ground water increases, the degree of attenuation tends to increase, leading to a decrease in pollution potential.

To identify a suitable value for depth to ground water in the model, eight landfills were monitored throughout the United States, and depths to the ground water below them were compiled (EPA, 1977). A typical depth to ground water of 5 m was observed among these landfills. A value of 0 m for Class I ground waters was chosen to model a worst-case depth to ground water. This conservative value, which represents a situation where the bottom of the landfill is occasionally or regularly below the water table, is used to provide the maximum protection to these most sensitive ground waters when determining allowable sludge contaminant concentrations. The 1-m value is used to represent the depth to all other ground waters, since this depth is believed to be the minimum distance necessary to provide adequate protection to human health and the environment while reflecting a reasonable worst-case scenario.

3.2.3 Aquifer Thickness

A value of 15 m was chosen as a reasonable aquifer thickness for three reasons. First, a very small thickness (under 3 m) will have influences in the model beyond simple mixing and dilution. In aquifers of this size, the leachate could constitute a large portion of the mixture, thereby nullifying one of the basic assumptions of the groundwater model, i.e., that leachate makes up a small portion of the total aquifer. Second, mixing does not increase with depth beyond about 15 m. Finally, the sensitivity analysis showed that when mixing beyond 15 m was modeled, no significant (i.e., order of magnitude) difference in the allowable contaminant levels in sludge were calculated. (See Section 4 for results and discussion of sensitivity analyses.)

3.2.4 Ground Water pH

The pH of a solution denotes the negative log of the hydrogen ion concentration of that solution. In general, the lower the pH, the greater the solubility of metals. The lowest pH examined in the MINTEQ simulations, 6 standard units (su), is therefore chosen as the most conservative value. (See discussion of MINTEQ in Section 2.)

3.2.5 Ground Water Eh

The measure of oxidation-reduction potential, Eh, indicates the potential of a solution to transfer electrons from the oxidant to the reductant. The higher the Eh value, the more readily metals migrate. The highest value of Eh examined in the MINTEQ simulations, 500 millivolts (mv), was chosen as the most conservative value.

3.3 UNSATURATED ZONE PARAMETERS

The unsaturated zone is the zone where the soil pore spaces are filled with water and air, distinguishing this zone from the saturated zone, where pore spaces are filled only with water. A number of soil types and conditions must be defined for the unsaturated zone as part of the transport modeling. These parameters are defined in the following sections.

3.3.1 Unsaturated Soil Type

Soil type in the unsaturated zone has a significant impact on the amount of recharge that can infiltrate into the ground and hence on the ability of a contaminant to move vertically into the aquifer. Additionally, the attenuation processes of filtration, biodegradation, sorption, and volitilization that can take place in the unsaturated zone depend on the soil type. The quantity of organic material (F_{oc}) present in the soil may also be an important factor for attenuation. In general, the pollution potential of a soil is largely affected by the type of clay present, the shrink/swell potential of that clay, and the grain size of the soil; that is, the less the clay shrinks and swells and the smaller the grain size of the soil, the less the pollution potential associated with the soil. Soil types in the unsaturated zone in order of increasing pollution potential are (1) nonshrinking clay, (2) clay loam, (3) silty loam, (4) loam, (5) sandy loam, (6) shrinking clay, (7) sand, (8) gravel, and (9) thin or absent soil (EPA, 1985a).

Sand has been selected as the modeled soil type for two reasons. First, Gerritse et al. (1982) used sand as a soil type when measuring the partitioning of elements between soil and a sewage sludge solution phase. These partitioning measurements (i.e., K_d values, or distribution coefficients) are considered the best available for the analysis of metal transport from landfilled sludge. The same soil type is also used for nonmetals for convenience and consistency of analysis. Second, sand is one of the soil types with the highest pollution potential and, as such, serves as a reasonable worst-case soil.

3.3.2 Unsaturated Zone Thickness

The unsaturated zone thickness is important in determining how long it takes contaminants to reach the aquifer, i.e., contaminant travel time. The longer the travel time, the greater the chance that attenuation processes may act on the contaminants. The degree of attenuation that occurs depends on (1) the length of time that the contaminant is in contact with the material through which it passes; (2) the grain size of the material through which it passes; (3) the physical and chemical characteristics of the material through which it passes; and (4) the distance that the contaminant has traveled. For most contaminants, a greater degree of attenuation is associated with longer travel times, greater media surface areas, and greater contaminant travel distances. In general, therefore, as the thickness of the unsaturated zone decreases, the pollution potential increases.

The thickness of the unsaturated zone over Class I ground water has thus been chosen as 0 m, a conservative but reasonable worst-case scenario for sensitive ground water. A value of 1 m has been chosen to represent the case of a landfill over other groundwater classes. This value is considered to represent a reasonable worst-case scenario for these less-sensitive groundwater types.

Only one unsaturated layer is assumed in these reasonable worst-case scenarios. Therefore, the thickness of the layer and the depth to ground water are the same and vary only with the underlying groundwater classification.

3.3.3 Slope of the Soil Moisture Retention Curve

The slope of the soil moisture retention curve is also called the slope of the curve plotting matric potential versus moisture content. The matric potential is a pressure potential that arises from the interaction of water with the matrix of solid particles in which it is embedded. It is also associated with water retention by the soil matrix. Water added to soil is subject to forces of capillary and surface adsorption that vary with water content (Marshall and Holmes, 1979).

The value of the slope of the soil moisture retention curve has been set at 4. This value is the typical value for sand, as can be seen in Table 3-3, which presents a range of slopes for the soil moisture retention curve.

3.3.4 Effective Porosity of the Unsaturated Zone

Porosity is the ratio of the void volume of a given soil or rock mass to the total volume of that mass. If the total volume is represented by V_T, the volume of the solids by V_S, and the volume of the voids by V_V, then the porosity, n, is defined as V_V/V_T. It is usually reported as a decimal fraction or as a percentage and ranges from 0 (no pore space) to 1 (no solid)

TABLE 3-3. **Typical Values for the Slope of the Soil Moisture Retention Curve by Soil Type**

Soil Texture	Slope
Clay	11.7
Silty clay	9.9
Silty clay loam	7.5
Clay loam	8.5
Sandy clay loam	7.5
Sandy silt loam	5.4
Silty loam	4.8
Sandy loam	6.3
Loamy sand	5.6
Sand	4.0

Source: Hall et al. (1977).

(Freeze and Cherry, 1979). The porosity of a uniform porous medium is largely a function of particle size. For soil types with small particle sizes such as clay, porosity increases to a maximum of around 50%. Porosities of coarser media like gravel decrease to a minimum of around 30%. These measured ranges of porosities suggest a strong correlation with mean particle diameter.

The term effective porosity refers to the amount of interconnected pore space available for fluid flow and is also expressed as a ratio of voids to total volume. The effective porosity is identical to porosity for many unconsolidated porous media and for many consolidated rocks (Todd, 1980).

The porosity value representative of coarse and medium sand is 0.39 m^3/m^3. For fine sand, this value is 0.43 m^3/m^3 (Todd, 1980). The lower value of 0.39 m^3/m^3 was chosen both as most representative of the modeled soil type and as most conservative.

3.3.5 Bulk Density in the Unsaturated Zone

The bulk density of soil is defined as the mass of dry soil divided by its total (or bulk) volume. Bulk density directly influences the retardation of solutes and is related to soil structure. In general, as soils become more compact, their bulk density increases. This relationship produces a dependency between porosity and bulk density. Freeze and Cherry (1979) note that the porosity is equal to one minus the ratio of bulk density to particle-size density. The particle density of soil materials varies over a very narrow range and can be fixed at a value of 2.65 g/cubic centimeter (cm^3) (Freeze and Cherry, 1979). If particle density is assumed equal to 2.65 g/cm^3, the bulk density can be expressed in terms of porosity as follows:

$$P_b = 2.65\,(1 - \theta) \tag{3}$$

where P_b = bulk density, g/cm^3
θ = porosity

For modeling purposes, bulk density values were obtained from the literature; the value chosen for the unsaturated zone, 1,400 kg/m^3, is a common value for sand.

3.3.6 Saturated Soil Hydraulic Conductivity

Unsaturated soil becomes saturated as water or leachate passes through it to the saturated zone below. Thus the hydraulic conductivity of the unsaturated zone must be estimated for those times when this zone becomes saturated. Hydraulic conductivity refers to the ability of the soil or aquifer materials to transmit water, which in turn controls the rate at which ground water will flow through soil or an aquifer under a given hydraulic gradient. The rate at which ground water flows also controls the rate at which a contaminant will move away from the point at which it enters the soil or aquifer. Hydraulic conductivity is governed by the amount and interconnection of void spaces in the soil or aquifer. These voids may occur as a consequence of intergranular porosity, fracturing, or bedding planes. In general, high hydraulic conductivities are associated with high pollution potential.

Saturated soil hydraulic conductivity (K_{sat}) is estimated from the representative values for saturated hydraulic conductivity in Freeze and Cherry (1979). The value chosen to represent a model landfill site is 10,000 m/yr, which is set at the high end of the hydraulic conductivity scale for clean sand.

3.4 SATURATED ZONE PARAMETERS

The saturated zone, with its water-filled pore spaces, is in most cases considered an aquifer. A number of parameters must be defined for the saturated zone when modeling pollutant transport. These parameters, which include soil type, soil hydraulic conductivity, effective porosity, hydraulic gradient, zone hydraulic conductivity, and bulk density, are discussed below.

3.4.1 Saturated Zone Soil Type

In the saturated zone, the soil type exerts the major control over the route and path length that a contaminant must follow. The path length is an important control (along with hydraulic conductivity and gradient) in determining the time available for attenuation processes such as sorption, reactivity, and dispersion. It is also important in determining the amount of effective surface area of the materials contacted by a contaminant in the aquifer. In general, larger grain sizes and more numerous fractures or openings within the aquifer are associated with higher permeabilities and lower attenuation capacity and, consequently, a greater pollution potential. Soil types in the saturated zone in order of increasing pollution potential are: (1) massive shale; (2) metamorphic/igneous; (3) weathered metamorphic/igneous; (4) bedded sandstone, limestone, and shale; (5) massive sandstone; (6) massive limestone; (7) sand and gravel; (8) basalt; and (9) Karst limestone (EPA, 1985a). For the same reasons as discussed in Section 3.3.1, sand has been chosen to represent a reasonable worst-case soil type.

3.4.2 Effective Porosity in the Saturated Zone

Effective porosities for general hydrogeologic classifications are presented in Table 3-4. The value of 0.1 m^3/m^3 has been chosen to represent

TABLE 3-4. Effective Porosities for General Hydrogeologic Classifications

Hydrogeologic Classification	Effective Porosity
Fractured crystalline silicates	0.01
Fractured and solutioned carbonates	0.10
Porous carbonates	0.10
Porous silicates	0.01
Porous unconsolidated silicates	0.16[a]
Fractured shale	0.01

[a] Average value.

Source: Shafer et al. (1984).

the effective porosity in the saturated zone. See the discussion of effective porosity in Section 3.3.4 for additional information on this parameter.

3.4.3 Hydraulic Gradient in Saturated Zone

The hydraulic gradient is, in general, a function of the local topography, the groundwater recharge volumes and locations, and the influence of withdrawals (e.g., well fields). It is also very likely to be indirectly related to porous media properties. Rarely are large gradients associated with very high conductivities. No functional relationship exists, however, to express this association.

The hydraulic gradient value selected for use in the model is 0.005 m/m and is based on an average value of a number of ground waters surveyed for the Hazardous Waste Management System Land Disposal Restrictions Regulation (51 FR 1602, January 14, 1986).

3.4.4 Saturated Zone Hydraulic Conductivity

The saturated zone hydraulic conductivity reflects the ease with which water is transported through porous media. For any given fluid, the hydraulic conductivity is a function of properties such as particle size, grain shape, connectivity, and tortuosity, which affect the porosity of the medium. Individual, site-specific measurements for hydraulic conductivity are usually difficult to make, and the spatial variability of "point" measurements is the subject of much current research.

In addition to porosity, saturated zone conductivity is related to the velocity of ground water and the hydraulic gradient. The velocity of ground water is a major determinant of solute transport in subsurface systems; in uniform, porous media, it is the dominant factor. Groundwater flow velocities vary widely. Mackay et al. (1985) report that velocities typically range between 1 to 100 m/yr. These ranges apply to typical "natural gradient" conditions, but higher velocities can exist under both induced situations (e.g., well-field drawdown) and extreme natural situations. For example, velocities in excess of 9,000 m/yr have been reported for a glacial outwash material (Guven et al., 1984).

Velocities are related to soil properties and other site-specific factors through Darcy's law. When Darcy's law and assumptions of steady flow in uniform, saturated media are used, the following expression for average pore velocity V is produced:

$$V = \frac{K_S \times S}{0} \qquad (4)$$

where V = velocity of ground water, m/yr
K_s = saturated zone hydraulic conductivity, m/yr
S = hydraulic gradient
0 = porosity

This equation can be rearranged as $K_s = 0V/S$ to derive the hydraulic conductivity. The saturated zone hydraulic conductivity was calculated based on a worst-case groundwater velocity of 100 m/yr, which was determined from the sensitivity analysis discussed in Section 4. Given a velocity, V, of 100 m/yr, a hydraulic gradient of 0.005 (see 3.4.4), and a saturated zone effective porosity of 0.1 m^3/m^3 (see 3.4.3), the saturated zone hydraulic conductivity for the model is calculated to be 2,000 m/yr.

3.4.5 Bulk Density in the Saturated Zone

Bulk density values are obtained from the literature and the value chosen for the saturated zone, 2,390 kg/m^3, is a common value for sand. More information on the derivation of bulk density is presented in Section 3.3.5.

3.5 SURFACE PARAMETERS

Several other miscellaneous parameters must also be defined for the model: the landfill site geometry, the distance to site boundaries, the surface wind velocity, the air temperature, the porosities of cover soil, and the cover thickness. These parameters and assigned values are discussed below.

3.5.1 Landfill Site Geometry

Table 3-5 presents the geometry used to model the landfill site. The landfill width and length represent the square root of the average area of landfills examined for the Hazardous Waste Management System Land Disposal Restrictions Regulation (51 FR 1602, January 14, 1986). The fill height and cell length are representative values for landfills.

3.5.2 Distance to the Property Boundary

Within the landfill property, the owner/operator can exercise control to ensure that there will be no exposure to pollutants at concentrations greater than the health effects levels. This area is thus known as the area of effective control. The area of effective control is considered to end at the point of potential exposure, which for Class II aquifer analysis is defined as the property boundary. The modeled pollutant concentrations do not exceed the health effects level at this point when the model has calculated the maximum allowable contaminant concentrations in the sludge.

To determine an appropriate distance that would represent the point of potential human exposure, the Agency surveyed 163 Part B hazardous waste land

TABLE 3-5. Landfill Site Geometry

Parameter	Value
Landfill width	100 m
Landfill length	100 m
Cell length	8 m
Fill height	3.46 m

disposal permit applications available to EPA as of October 1984. Based on this information, the Agency believes that 150 m is a reasonable, conservative estimate of the point of effective potential human exposure.

3.5.3 Surface Wind Velocity

A value of 1 m/sec has been selected to represent a reasonable value for surface wind velocity. For site-specific modeling, more exact calculations may be made with values obtained from local weather stations or measured on site (Tucker and Preston, 1984).

3.5.4 Air Temperature

An air temperature of 15°C has been selected as a reasonable value. The value could be measured on site or taken from local weather stations for a more exact calculation when site-specific modeling is performed.

3.5.5 Air-filled Porosity of Cover Soil

It is assumed that cover soils will be drained to field capacity, which is the water content found when a thoroughly wetted soil has drained for about 2 days. Under this assumption, the air-filled porosity is set equal to the effective porosity, or 0.1 m^3/m^3.

3.5.6 Total Porosity of Cover Soil

A value of 0.4 m^3/m^3 has been chosen as representative of a soil cover of sand, a worst-case scenario.

3.5.7 Cover Thickness

The thickness of the active cover has been set at 0.3 m; the thickness of the final cover has been set at 1 m. These values are minimal values and are used to represent worst-case conditions.

3.6 CHEMICAL-SPECIFIC PARAMETERS

A number of chemical-specific factors must be quantified before being input to the model. These factors include the concentration of contaminants in the sludge by dry-weight basis; the concentration of contaminants in the leachate; the distribution coefficient; the saturated and unsaturated zone decay rate; the background concentration of contaminants, and other factors. The chemical-specific parameters and the values assigned to them are discussed below.

3.6.1 Concentrations of Contaminants in Sludge

The concentrations of contaminants in the modeled sludge are presented in Table 3-6. In many cases, the values were taken from 95th percentile values obtained from a survey of 40 POTWs (EPA, 1982). The values used for chlordane, lindane, DDT/DDD/DDE, trichloro-ethylene, and toxaphene, however, were the maximum values reported. Other values were derived as noted in the table. These values were used to begin the iterative runs as a first approximation (see Section 2).

3.6.2 Concentrations of Contaminants in Sludge Leachate

The concentrations of contaminants in the modeled sludge leachate are presented in Table 3-7. Most of these values were derived using the modeled

TABLE 3-6. Concentrations of Contaminants in Sludge (dry weight)

Parameter	Concentration (mg/kg)
Arsenic	20.75
Cadmium	88.13[a]
Copper	5600.[a]
Lead	1070.
Mercury	5.85
Nickel	920.
Benzene	6.6
Benzo(a)pyrene	1.935
Bis(2-ethylhexyl)phthalate	459.
Chlordane	12.
DDT/DDD/DDE	0.930
Dimethylnitrosamine	0.272[b]
Lindane	0.220
Polychlorinated biphenyls	2.90
Trichloroethylene	17.85
Toxaphene	10.8

[a] The value was obtained by averaging the maximum values reported by a number of surveys.

[b] The value was the mean value reported in EPA (1982).

Source: EPA (1982), except as noted.

TABLE 3-7. Concentrations of Contaminants in Sludge Leachate

Parameter	Concentration (mg/L)
Arsenic	1.[a]
Cadmium	0.2[b]
Copper	37.[b]
Lead	10.[b]
Mercury	0.69[c]
Nickel	3.4[d]
Benzene	0.12
Benzo(a)pyrene	0.000006
Bis(2-ethylhexyl)phthalate	25.[a]
Chlordane	0.00014
DDT/DDD/DDE	0.01[e]
Dimethylnitrosamine	0.014
Lindane	0.00039
Polychlorinated biphenyls	0.0000018
Trichloroethylene	0.022
Toxaphene	0.15

[a] The value is based on an estimated liquid concentration that requires at least one year to deplete the mass of the leachate in the sludge.

[b] The value was the maximum reported for leachate from a sludge monofill (EPA, 1978).

[c] The value was recommended by Betsy Southerland (1987).

[d] The value was the maximum effluent reported for municipal wastewaters. (Barth et al., 1965).

[e] The value was the maximum effluent value reported (EPA, 1978).

Source: Calculated from K_{oc} and the values in Table 3-6, except as noted.

sludge concentrations in Table 3-6 and the organic carbon content of the soil (K_{oc}) at 5% total solids and 50% organic solids. The sources of other values are as noted in the table. These values were used to begin the iterative runs as a first approximation.

3.6.3 Distribution Coefficient

Contaminant transport in soil systems is directly related to contaminant/soil interactions. The affinity that soil particles have for contaminants may result from ion exchange on charged sites or adsorption due to surface forces. When that capacity is exceeded, soluble contaminants will move through the soil at the same velocity as the bulk leachate. The affinity between a soil and a contaminant, and therefore a soil's capacity to hold a contaminant, is characterized by the distribution coefficient K_d. Representative K_d values are defined as the equilibrium ratio of the contaminant concentration in soil (mg/kg) to that in associated water (mg/L). Table 3-8 presents the values used to represent the distribution coefficients in the model.

When a soil's capacity to contain a contaminant is exhausted, the contaminant can be considered to have a K_d value of zero and will move unimpeded with the leachate until it reaches soil with residual capacity. Hence, the capacity of the soil column (SC) between the landfill and the point of compliance (the landfill boundary) must be compared to the total contaminant available in the sludge (M). If M is greater than SC, then the risk calculations must be made with $K_d = 0$; whereas if M is less than SC, then the measured values of K_d from the literature should be applied.

For organics, the K_d concept is more broadly useful because adsorption accounts for most soil attenuation. In this case, K_d is calculated as follows:

TABLE 3-8. Distribution Coefficients, K_d

Parameter	Coefficient (L/kg)
	Inorganic (K_d)
Arsenic	5.86
Cadmium	14.9
Copper	41.9
Lead	234.0
Mercury	322.0
Nickel	12.2
	Organic (K_d)[a]
Benzene	0.0074
Benzo(a)pyrene	550.0
Bis(2-ethylhexyl)phthalate	200000.
Chlordane	17.0
DDT/DDD/DDE	500.0
Dimethylnitrosamine	0.000004
Lindane	0.108
Polychlorinated biphenyls	32.0
Trichloroethylene	0.0198
Toxaphene	0.06

[a] Based on an f_{oc} of 10^{-4} x K_{oc} (see text).

$$K_d = (K_{oc})(f_{oc}) \qquad (5)$$

where K_{oc} = organic carbon content of the soil
f_{oc} = fraction of soil consisting of organic matter

If the values for K_{oc} have not been determined experimentally, equations are available to estimate them from octanol-water partition coefficient data, known as K_{ow} or solubility.

For organic contaminants, K_d is a function of organic content in soil. The value of 10^{-4} was selected for the f_{oc} values as a typical value for sand. It is assumed that subsoils in the aquifer will not contain organic matter and, therefore, the K_d for organics in the saturated zone is set equal to zero.

3.6.4 Saturated and Unsaturated Zone Decay Rate

Reductions in pollutant concentrations due to degradation processes such as hydrolysis and biochemical oxidation are characterized by a degradation constant, or decay rate, which is related to the time required for the contaminant concentration to be reduced to one-half its initial value. (See Section 2 for more details.) The values used in the model to represent the saturated and unsaturated zone decay rates are presented in Table 3-9.

3.6.5 Background Concentrations of Contaminants

The background concentrations of contaminants in ground water were taken from "Significance of Properties and Constituent's Reported in Water Analysis" (Hen, 1985) and are presented in Table 3-10. Where explicit values were not given, values were estimated from this source. Since metals are ubiquitous,

TABLE 3-9. Saturated and Unsaturated Zone Decay Rates

Parameter	Decay Rate (yr^{-1})
Arsenic	0.0
Cadmium	0.0
Copper	0.0
Lead	0.0
Mercury	0.0
Nickel	0.0
Benzene	0.0
Benzo(a)Pyrene	0.0
Bis(2-ethylhexyl)phthalate	0.0
Chlordane	8.43[a]
DDT/DDD/DDE	0.904[b]
Dimethylnitrosamine	607.0[a]
Lindane	8.43[c]
Polychlorinated biphenyls	0.0
Trichloroethylene	0.904[d]
Toxaphene	0.0

[a] This value was obtained from EPA's Athens Environmental Research Laboratory (Kollig).

[b] This value was taken from Wolf et al. (1977).

[c] This value was taken from Raghu and MacRae (1966).

[d] This value was taken from Kleopfer (1985).

Sources: As noted above.

TABLE 3-10. Background Concentrations of Contaminants

Parameter	Concentration (mg/L)
Arsenic	0.0004
Cadmium	0.001
Copper	0.01
Lead	0.001
Mercury	0.0003
Nickel	0.0027
Benzene	0.0
Benzo(a)pyrene	0.0
Bis(2-ethylhexyl)phthalate	0.0
Chlordane	0.0
DDT/DDD/DDE	0.0
Dimethylnitrosamine	0.0
Lindane	0.0
Polychlorinated biphenyls	0.0
Trichloroethylene	0.0
Toxaphene	0.0

they can be expected to be found naturally in ground waters. Organics, however, are not expected to be found in noncontaminated ground waters and, therefore, the background concentrations of organics were set to zero.

3.6.6 Health Effects Levels

The health effects level (HEL, in mg/L) is defined as a groundwater concentration used to evaluate the potential for adverse effects on human health as a result of sludge landfilling. The monofill pollutant limits that represent allowable contaminant concentrations in sludge to be monofilled, which are presented in §503.43 of the Sewage Sludge Technical Regulations, were calculated to result in groundwater concentrations below the HEL at the point of compliance. The point of compliance is the property boundary for Class II aquifer analysis, but is the point at which leachate enters the aquifer for Class I aquifer analysis. (See Section 2.3 for a definition of these two types of analyses.)

The HEL values, presented in Table 3-11, are set equal to MCLs where they are available. Where MCLs are not available, a HEL is calculated based on RfDs for threshold-acting contaminants (i.e., noncarcinogens) or q_1* values for carcinogens. The RfD is a benchmark dose for noncarcinogens derived from the no-observed-adverse-effects level (NOAEL) by applying generally order-of-magnitude uncertainty factors (UFs). These UFs reflect various types of data used to estimate RfDs and an additional modifying factor (MF), which is derived based on professional judgment following review of the contaminant's entire data base. In general, the RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is very likely to result in no appreciable risk of health effects during a lifetime. The RfD is appropriately expressed in units of mg/kg-body weight (bw)/day. The HEL was calculated from the RfD using the following equation:

TABLE 3-11. Health Effects Levels (mg/L)

Parameter	Concentration (mg/L)
Arsenic	0.05[a]
Cadmium	0.01[a]
Copper	1.3[a]
Lead	0.05[a]
Mercury	0.002[a]
Nickel	1.75[b]
Benzene	0.005[a]
Benzo(a)pyrene	0.00003[c]
Bis(2-ethylhexyl)phthalate	0.0248[c]
Chlordane	0.00021[c]
DDT/DDD/DDE	0.00102[c]
Dimethylnitrosamine	0.00001[c]
Lindane	0.004[a]
Polychlorinated biphenyls	0.0000454[c]
Trichloroethylene	0.005[a]
Toxaphene	0.0005[a]

[a] The value is a Maximum Contaminant Level (MCL) under the Safe Drinking Water Act (SDWA).

[b] The value is calculated from a risk reference dose (RfD) established by EPA's Office of Research and Development.

[c] The value is calculated from a q^1 value representing a maximum allowable dose for carcinogens at the 10^{-5} level.

Sources: As noted above.

$$HEL = \frac{RfD \times BW}{I_w} \quad (6)$$

where RfD - reference dose (mg/kg/day)
BW - human body weight (kg)
I_w - water ingestion rate (L/day)

The human body weight was set at 70 kg and the water ingestion rate at 2 L/day for adults, standard values used by EPA.

For most carcinogenic chemicals, the linearized multistage model is recommended for estimating human cancer potency from animal data (49 FR 46298). When epidemiological data are available, potency is estimated based on the observed relative risk in exposed versus nonexposed individuals and on the magnitude of exposure. Guidelines for using these procedures have been presented in the Federal Register (45 FR 79350-47353; 50 FR 46294-46301) and in each of a series of health assessment documents prepared by the Office of Health and Environmental Assessment (OHEA). The potency value, q_1* normally used in risk assessments, is the upper-bound estimate of the slope of the dose-response curve in the low-dose range. This value is expressed in terms of risk per dose, where dose is expressed in units of mg/kg/day. Thus, q_1* is expressed in units of mg/kg/day^{-1}. The HEL was calculated from q_1* using the following equation:

$$HEL = \frac{(RL \times BW)/q_1^*}{I_w} \quad (7)$$

where RL - risk level
BW - human body weight (kg)
I_w - water ingestion rate L/day)

As in the RfD computation, the human body weight was set at 70 kg and the water ingestion rate at 2 L/day for adults; the risk level (q_1*) was set at 10^{-5}.

3.6.7 Molecular Weight

The values presented in Table 3-12 are standard molecular weights for several contaminants of concern. These weights are used in the vapor loss simulations and have been "hard-wired" into the model.

3.6.8 Henry's Law Constants

Henry's law constants are used to calculate the vapor loss of contaminants from sludge and are "hard-wired" into the model. Table 3-13 presents constants for several contaminants of concern. The values for benzo(a)pyrene and bis(2-ethylhexyl)phthalate were calculated from vapor pressure and solubility data that are maintained in the Henry's law database at Research Triangle Park, using the procedures in Mackay and Shui (1981). The vapor pressure of bis(2-ethylhexyl)phthalate was calculated from boiling points at various pressures using Antoine's equation and the solubility data was obtained from Verschueran (1983). The value for chlordane was also obtained from the database, but was taken from Hwang (1982). Values for DDT/DDE/DDD and polychlorinated biphenyls (PCBs) were taken from Lyman et al (1982). Finally, the value for dimethylnitrosamine was taken from Dawson et al. (1980).

TABLE 3-12. Standard Molecular Weights for Modeled Pollutants

Contaminant	Molecular Weight
Benzo(a)pyrene	252.32
Bis(2-ethylhexyl)phthalate	391
Chlordane	410
DDT/DDE/DDD	354.5
Dimethylnitrosamine	74.08
Polychlorinated biphenyls	300

TABLE 3-13. Henry's Law Constants for Selected Contaminants

Contaminant	Constant
Benzo(a)pyrene	0.000000017
Bis(2-ethylhexyl)phthalate	0.004625
Chlordane	0.002
DDT/DDE/DDD	0.0017a
Dimethylnitrosamine	0.02b
Polychlorinated biphenyls	0.16a

Sources: As noted in text.

SECTION FOUR

SENSITIVITY ANALYSIS

In developing the risk assessment methodology for each sludge disposal option, the Agency selected or created mathematical models that would predict the average long-term human health and environmental effects of continually disposing of sludge in the soil and/or water. The SLUDGEMAN model was developed to predict the maximum allowable pollutant concentrations in sewage sludge at a landfill site that would not detrimentally affect the MEI. The MEI was defined as a person who drinks 2 L of ground water per day from the site over a 70-yr lifespan. This model or a modification thereof will probably be used for the surface disposal option because the movement of pollutants in the soil and into the ground water is similar for both practices. The air modeling results, however, will be different for the two models because no cover is maintained on a surface disposal site.

Although no sensitivity analysis was performed for the surface disposal option, similarity to the monofilling option was assumed in evaluating the potential for damage to human health and the environment from surface disposal. The following discussion reviews the results of the monofilling sensitivity analysis. Because no cover is maintained on surface disposal sites, the air effects from surface disposal may exceed those from monofilling. The air modeling results are reported indirectly here because they were taken into account in SLUDGEMAN, and the groundwater results were adjusted to reflect the air effects. This influence should be taken into account when evaluating the results of the monofilling sensitivity analysis for the surface disposal option.

Existing disposal practices were extensively reviewed to develop reasonable worst-case values for the site characteristics used in SLUDGEMAN. Six sets of sensitivity analyses were performed on selected model parameters and on the definition of the MEI to evaluate whether valid worst-case assumptions were used. The results of these analyses are discussed below and are detailed in Appendix B of the Technical Support Document for Monofilling of Sewage Sludge (EPA, 1989b).

The eight parameters evaluated for their effect on allowable sludge concentrations are:

- Depth of the soil layer between the landfill and ground water
- Organic carbon fraction of that soil layer
- Distance to the point of compliance
- Groundwater velocity
- Aquifer thickness
- Net recharge rate
- Eh and pH conditions of the ground water
- Moisture content of the sludge

Adsorption and decay coefficients -- parameters specific to each chemical -- were not evaluated in the sensitivity analyses. The depth of the intervening soil layer, the net recharge rate, and the aquifer thickness proved to be the most sensitive parameters.

Initially, a baseline case scenario was run, against which all subsequent runs were compared. The input parameters for the baseline case are presented in Table 4-1.

The sensitivity analyses were performed by holding all parameters at their baseline value or at other specific values, except the parameter being evaluated. One parameter, distance to the property boundary, was varied in all the sensitivity analyses to determine the effect of this variable in combination with all the other parameters of interest in these analyses.

Because parameter sensitivity depends on the baseline conditions selected

TABLE 4-1. Input Parameters for the Baseline Case

Parameter	Volume
Depth to ground water	0 m
Net recharge rate	0.5 m/yr
Groundwater gradient	0.005 m/m
Hydraulic conductivity of saturated zone	2,000 m/yr
Groundwater velocity	100 m/yr
Aquifer mixing thickness	15 m
Eh and pH	+500 mv/6.0
Sludge moisture content	0.80 kg/kg
Sludge storage capacity	0.90 kg/kg
Sludge density	1,025 kg/m^3
Width of landfill	100 m
Length of landfill	100 m
Height of fill	3.46 m
Aquifer width	1,000 m

and on the adsorption and decay coefficients assigned, the analysis may not give a generally applicable measure of sensitivity, but rather a specific measure applicable to the testing conditions. It is also important to note that the validity of the sensitivity analysis hinges on the validity of the model framework itself.

4.1 EFFECT OF SOIL DEPTH AND CARBON CONTENT

The first set of sensitivity analyses evaluated the effect on maximum allowable sludge pollutant concentrations of (1) the depth of the soil layer between the landfill and the ground water and (2) the organic carbon content of that soil layer. The organic content of the soil was examined because it influences the amount of organic pollutants that will adsorb onto the soil, thus not reaching the aquifer. It does not, however, affect metals leaching to ground water. Depth to ground water proved to be an extremely sensitive parameter for DDT, lindane, and bis(2-ethylhexyl)phthalate (BEHP), as will be discussed in the following sections.

4.1.1 Results for Class I Aquifers

For Class I ground waters, as described in Section 2, human health criteria must be met at the point where the leachate first enters the aquifer. Thus, the location of the landfill property boundary is not a factor in the analysis. Table 4-2 presents the results of the analyses for Class I aquifers. When a low soil organic carbon content of 0.0001 is assumed, as depth to ground water increases from 0 to 1 meter (m), the maximum allowable concentrations increase for all the regulated pollutants by less than an order of magnitude. For soils with a high organic carbon content of 0.01, as depth to ground water increases from 0 to 1 m, the allowable sludge concentrations increase by two orders of magnitude for DDT and by three orders of magnitude for lindane. In other words, leachate from landfills with sludges containing

TABLE 4-2. Effect of Soil Depth and Organic Carbon Content on Maximum Allowable Sludge Pollutant Concentrations for Class I Aquifers

Depth of Soil Between Landfill and Aquifer	Soil Organic Carbon Content	Pollutant	Orders of Magnitude of Increase in Maximum Allowable Sludge Pollutant Concentrations over Baseline
1-m	0.0001	All regulated pollutants	Less than 1
1-m	0.01	DDT	2
		Lindane	3
5-m	0.0001	DDT	1
5-m	0.01	BEHP	1
		DDT	2
		Lindane	3
10-m	0.0001	DDT	1
		Lindane	1
10-m	0.01	BEHP	1
		DDT	3
		Lindane	3

the maximum allowable concentrations of these pollutants calculated by these sensitivity analyses would still meet the human health criteria at the point where the leachate first enters the aquifer.

For soils with an organic content of 0.0001, as the depth to ground water increases to 5 m, allowable sludge concentrations for DDT increase by one order of magnitude. When an organic soil content of 0.01 m is assumed, as depth to ground water increases to 5 m, allowable sludge concentrations increase by one order of magnitude for BEHP, two orders of magnitude for DDT, and three orders of magnitude for lindane.

For soils with an organic content of 0.0001, as the depth to ground water increases to 10 m, the allowable sludge concentrations for DDT and lindane increase by one order of magnitude. When an organic content of 0.01 is assumed, the 10-m depth increases allowable sludge concentrations by one order of magnitude for BEHP and by three orders of magnitude for DDT and lindane.

4.1.2 Results for Class II Aquifers

For Class II ground waters, human health criteria must be met after the leachate mixes with the ground water and is transported to the property boundary of the landfill where water supply wells might be constructed. The property boundary of the landfill is, therefore, an important parameter in the analysis of Class II aquifers. The greater the distance from the edge of the waste management unit to the property boundary, the greater the allowable sludge concentrations because pollutants undergo more dilution and decay the farther they travel in the aquifer.

The sensitivity analyses of landfill model parameters were performed assuming property boundary locations at 50 m, 150 m, 500 m, and 1,000 m from the landfill. The results are given in Table 4-3. For the 50-, 150-, and 500-m boundaries, a 0.0001 soil organic carbon content caused an order-of-

TABLE 4-3. Effect of Soil Depth and Organic Carbon Content on Maximum Allowable Sludge Pollutant Concentrations for Class II Aquifers

Depth of Soil Between Landfill and Aquifer	Location of Property Boundary	Soil Organic Carbon Content	Pollutant	Orders of Magnitude of Increase in Maximum Allowable Sludge Pollutant Concentrations over Baseline
1-m and 5-m	50-m, 150-m and 500-m	0.0001	DDT	1
1-m and 5-m	50-m, 150-m and 500-m	0.0001	All regulated pollutants other than DDT	Insignificant
1-m and 5-m	50-m, 150-m and 500-m	0.01	DDT BEHP	1 1
1-m and 5-m	50-m	0.01	Lindane	3
	150-m	0.01	Lindane	3
	500-m	0.01	Lindane	1
10-m		0.0001	DDT Lindane	1 1
10-m		0.01	BEHP DDT	1 1
1-m	1000-m	0.01 0.0001	All regulated pollutants	Insignificant
5-m and 10-m	1000-m	0.0001	All regulated pollutants	Insignificant
5-m and 10-m	1000-m	0.01	BEHP	1
5-m and 10-m	1000-m	0.01	All regulated pollutants other than BEHP	Insignificant

magnitude increase in only one pollutant, DDT, at both the 1- and 5-m soil depths, compared to the baseline of 0-m. When a 0.01 soil organic carbon content was assumed, the maximum allowable pollutant concentrations of three pollutants, DDT, BEHP, and lindane, increased significantly at both the 1- and 5-m soil depths for all three boundaries. Significant increases also occurred for these three pollutants at the 10-m soil depth.

The sensitivity analyses indicate that the 1,000-m property boundary parameter was very insensitive to the depth of the intervening soil layer. The 1,000-m distance is so far from the landfill that allowable sludge concentrations are controlled by dilution and transformation processes in the aquifer rather than by processes in the soil layer under the landfill. A 1-m-deep soil layer caused no significant increase in maximum allowable pollutant concentrations over reasonable worst-case sludge concentrations. The 5- and 10-m-deep soil layers caused an order-of-magnitude increase in just one pollutant, BEHP, and that increase only occurred when the soil was modeled with a high fraction (0.01) of organic carbon content.

4.2 EFFECT OF RECHARGE RATE

The second set of sensitivity analyses evaluated the effect of recharge rates on maximum allowable sludge concentrations. The net recharge rate is the amount of rain water that seeps through the landfill and the underlying soil into the ground water. It is calculated as the average annual amount of rainfall minus losses from evaporation to the air and transpiration by plants. Recharge rate does not affect the maximum allowable pollutant concentrations in sludge for Class I ground waters. For the Class I analysis, the leachate itself must meet human health criteria, thus the rate at which the leachate is added to ground water is not important. For the Class II analysis, however, the recharge rate has proved to be an extremely sensitive parameter.

The highest national average recharge rate (0.5 m/yr) was assumed for the baseline case. A high value was chosen because the higher the recharge rate, the more rapidly the pollutants are flushed into the ground water. The sensitivity analyses examined lower net recharge values of 0.25 m/yr and 0.00635 m/yr. The 0.25 m/yr recharge rate did not cause a large increase in the allowable sludge concentrations; the increase was only a factor of two to three whether the property boundary was set at 50 m, 150 m, 500 m, or 1,000 m. In contrast, the very low (0.00635 m/yr) recharge rate caused order-of-magnitude increases in allowable sludge concentrations for both metals and organics. For Class II ground waters at 50-m and 150-m boundaries, the 0.00635-m/yr recharge rate increased allowable sludge concentrations by three orders of magnitude for arsenic and cadmium and by one order of magnitude for the other pollutants. For Class II ground waters at 500-m and 1,000-m boundaries, the 0.00635-m/yr recharge rate increased allowable sludge concentrations by two orders of magnitude for cadmium and one order of magnitude for the other pollutants.

4.3 EFFECT OF GROUNDWATER VELOCITY

The third set of sensitivity analyses evaluated the effect of groundwater velocity on maximum allowable sludge concentrations. Groundwater velocity is a function of gradient and hydraulic conductivity of the groundwater zone. The baseline groundwater velocity is 100 m/yr based on a gradient of 0.005 m/m and a hydraulic conductivity of 2,000 m/yr. The sensitivity analysis examined the effect of raising the groundwater velocity to 1,000 m/yr (based on a gradient of 0.1 m/m and hydraulic conductivity of 1,000 m/yr) and lowering it to either 10 m/yr (based on a gradient of 0.005 m/m and hydraulic conductivity of 200 m/yr) or 1 m/yr (based on a gradient of 0.0001 m/m and hydraulic conductivity of 1,000 m/yr). Table 4-4 presents the results of these analyses.

The sensitivity analyses demonstrated that high groundwater velocity (1,000 m/yr) tends to increase allowable sludge concentrations because it provides greater dilution of the contaminants as they enter the aquifer. For

TABLE 4-4. Effect of Groundwater Velocity on Maximum Allowable Sludge Concentrations

Groundwater Velocity	Location of Property Boundary	Pollutant	Orders of Magnitude of Increase in Maximum Allowable Sludge Pollutant Concentrations over Baseline
1,000 m/yr	150-m	Cadmium	3
		Arsenic	3
10 m/yr	50-m	DDT	1
		TCE	2
		Lindane	2
10 m/yr	150-m	DDT	1
		TCE	3
		Lindane	3
1 m/yr	50-m	DDT	1
		TCE	3
		Lindane	3
1 m/yr	150-m	DDT	1
		TCE	3
		Lindane	3
10 m/yr	500-m	DDT	1
		TCE	1
		Lindane	1
1 m/yr	500-m	DDT	1
		TCE	1
		Lindane	1
10 m/yr	1000-m	All pollutants	Insignificant
1 m/yr	1000-m	All pollutants	Insignificant

degradable organics, however, increased dilution at higher velocities may be offset by a decrease in travel time to the property boundary, which reduces the time in which decay can occur before the sludge contaminants reach the boundary.

Groundwater velocity was generally a relatively insensitive parameter for all pollutants except cadmium and arsenic. For Class I ground waters, the variation in groundwater velocity has no effect on maximum allowable sludge concentrations because human health criteria must be met at the point of leachate entry into the aquifer, before the leachate is transported in the aquifer. For Class II ground waters, the high groundwater velocity of 1,000 m/yr caused a large increase in the allowable sludge concentrations for cadmium and arsenic independent of property boundary location (50 m, 150 m, 500 m, or 1,000 m). The high velocity also increased the allowable sludge concentrations for the other regulated pollutants, but by lesser amounts. At the 50-m boundary, the high groundwater velocity caused a three-order-of-magnitude increase in the allowable sludge concentrations for arsenic and cadmium. At the 150-m boundary, a one-order-of-magnitude increase in velocity (i.e., 1,000 m/yr) caused a three-order-of-magnitude increase in the allowable arsenic and cadmium sludge concentrations. This increase is much greater than that which can be explained by increased dilution.

The lower groundwater velocities of 10 m/yr and 1 m/yr also increased the allowable sludge concentrations over those calculated for the baseline situation. The most significant increases were for DDT, lindane, and trichloroethylene at landfills with 50-m, 150-m, and 500-m boundaries. At the 1,000-m boundary, the lower velocities did not cause significant increases in allowable sludge concentrations for any pollutants. The effect of groundwater velocity was reduced at 500 m and 1,000 m because most of the chemical was already well dispersed and degraded by the time it reached those distances, no matter what the velocity.

4.4 EFFECT OF AQUIFER THICKNESS

The fourth set of sensitivity analyses evaluated the effect of aquifer thickness, or the depth of an aquifer, on maximum allowable sludge concentrations. The greater the aquifer thickness, the greater the volume of water into which the leachate plume may diffuse. Aquifer thickness proved to be a somewhat sensitive parameter.

The reasonable worst-case situation is based on a relatively small aquifer thickness of 15 m. The sensitivity analyses looked at aquifers 78.6 m and 560 m in depth, as well as at a smaller aquifer of 5 m in depth. Aquifer thickness does not affect allowable sludge concentrations for Class I ground waters because human health criteria must be met before the water in the aquifer dilutes the contaminants. For Class II ground waters where the property boundary was assumed to be at 50 m, 150 m, or 500 m, the two greater aquifer thicknesses (78.6 m and 560 m) caused no significant increase in allowable sludge concentrations. The smaller thickness (5 m), however, did decrease the allowable sludge concentration by an order of magnitude.

When the property boundary was assumed to be located at 1,000 m, aquifer thickness significantly affected the allowable sludge concentrations. The 1,000-m distance is so far from the landfill that allowable sludge concentrations are controlled by dilution and transformation processes in the aquifer rather than by processes in the soil layer under the landfill. At the 1,000-m boundary, the two thicker aquifers increased allowable sludge concentrations by an order of magnitude for arsenic and two orders of magnitude for cadmium. The thinner aquifer decreased allowable sludge concentrations by one order of magnitude.

4.5 EFFECT OF Eh AND pH

The fifth set of sensitivity analyses examined the effect of ground water Eh and pH on maximum allowable sludge concentrations. Eh denotes the potential required to transfer electrons from the oxidant to the reductant, and pH denotes the negative log of the hydrogen ion concentration. At the point where leachate enters the ground water, the landfill model reduces groundwater metal concentrations to the solubility limit of the metal. The amount of metal that precipitates out of solution depends on the Eh and pH of the ground water.

Ground water with a high Eh and a low pH will keep most metals in the dissolved state -- a more hazardous situation -- and will thus require more stringent controls on metal concentrations in sludge. The reasonable worst-case values are an Eh of +500 millivolts (mv) and a pH of 6 standard units (su). Ground water with a low Eh and a high pH is more likely to cause metals to precipitate out of solution, so higher metal concentrations can be allowed in the sludge.

The sensitivity analyses examined the effect of a moderate Eh condition of +150 mv and a pH of 6, as well as an Eh condition of -200 mv and a pH of 7. Only the latter Eh and pH conditions had much effect on allowable sludge concentrations.

For Class I ground waters, an Eh of -200 mv and a pH of 7 caused an increase in the allowable sludge concentrations of three orders of magnitude for arsenic, one order of magnitude for copper, and two orders of magnitude for lead and nickel. For Class II ground waters at a 50-m boundary, this condition caused a three-order-of-magnitude increase in sludge concentrations for arsenic and a one-order-of-magnitude increase for lead. For Class II ground waters at a 150-m and 500-m boundary, it caused a two-order-of-magnitude increase for arsenic, and at 1,000-m a one-order-of-magnitude increase for arsenic.

4.6 EFFECT OF SLUDGE MOISTURE CONTENT

The sixth and final set of sensitivity analyses evaluated the effect of sludge moisture content on maximum allowable sludge concentrations. The reasonable worst-case value for sludge moisture content is 80%. Sludges with higher moisture content were expected to increase allowable sludge concentrations because they contain a smaller proportion of contaminated solids. Sludges with lower moisture content were expected to decrease allowable concentrations because they contain a higher proportion of contaminated solids. The sensitivity analyses examined the effect of a higher moisture content (95%) and a lower moisture content (60%).

Sludge moisture content proved to be a relatively insensitive parameter. For Class I ground waters, use of a 95% moisture sludge did not significantly increase the allowable sludge concentrations and use of a 60% moisture sludge did not significantly decrease the allowable sludge concentrations. For Class II ground waters at 50 m, 150 m, 500 m, and 1,000 m, the analysis showed that use of a 95% moisture sludge would increase the allowable sludge concentrations by an order of magnitude, whereas use of a 60% moisture sludge would decrease allowable sludge concentrations by less than an order of magnitude.

SECTION FIVE

POLLUTANT LIMITS

The proposed 405(d) sewage sludge regulations are based on maximum pollutant limits that have been calculated for each sludge pollutant evaluated for this regulation. For SLUDGMAN, the Agency selected reasonable worst-case values for monofill site characteristics (detailed in Section 3) to calculate these maximum allowable sludge concentrations (pollutant limits) using the methodology in Section 2. Given that surface disposal sites are assumed to be relatively small, the Agency has determined that there is little potential for human exposure to contaminants from surface disposal of sludge. The sludge pollutant concentrations are, therefore, limited to the less stringent of the following:

- 98th percentile concentrations reported in the EPA 40-Cities Study (EPA, 1982)
- Pollutant concentrations derived from the monofilling risk assessment

The 98th percentile concentrations for the pollutants measures are listed in Table 5-1; the pollutant concentrations from the monofilling assessment are listed in Table 5-2.

Figure 5-1 shows how pollutant limits, pathogen and vector attraction reduction requirements, and management practices determine whether sewage sludge may be surface disposed.

TABLE 5-1. 98th Percentile Concentrations (mg/kg)

	Maximum Sewage Sludge Concentration (mg/kg)
Benzo(a)pyrene	99
Bis(2-ethylhexyl) phthalate	782
Cadmium	385
DDT/DDE/DDD (total)[b]	0.95
Lead	1,622
Lindane	2.3
Mercury	17
Trichloroethylene	181
Toxaphene	0.5

[a] Dry-weight basis.

[b] DDT -- Bis 1,1-(4-chlorophenyl)-2,2,2-trichloroethane
DDE -- Bis 1,1-(4-chlorophenyl)-2,2-dichloroethylene
DDD -- Bis 1,1-(4-chlorophenyl)-2,2-dicloroethane

TABLE 5-2. Pollutant Concentrations from Monofill Analysis

	Maximum Sewage Sludge Concentration (mg/kg)
Arsenic	36
Benzene	15
Benzo(a)pyrene	250
Bis(2-ethylhexyl) phthalate	1600
Cadmium	380
DDT/DDE/DDD (total)[b]	51
Dimethylnitrosamine	1.4
Lead	1,600
Lindane	75
Mercury	26
Polychlorinated biphenyls	49
Trichloroethylene	180
Toxaphene	1.1

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[a] Dry-weight basis.

[b] DDT -- Bis 1,1-(4-chlorophenyl)-2,2,2-trichloroethane
DDE -- Bis 1,1-(4-chlorophenyl)-2,2-dichloroethylene
DDD -- Bis 1,1-(4-chlorophenyl)-2,2-dicloroethane

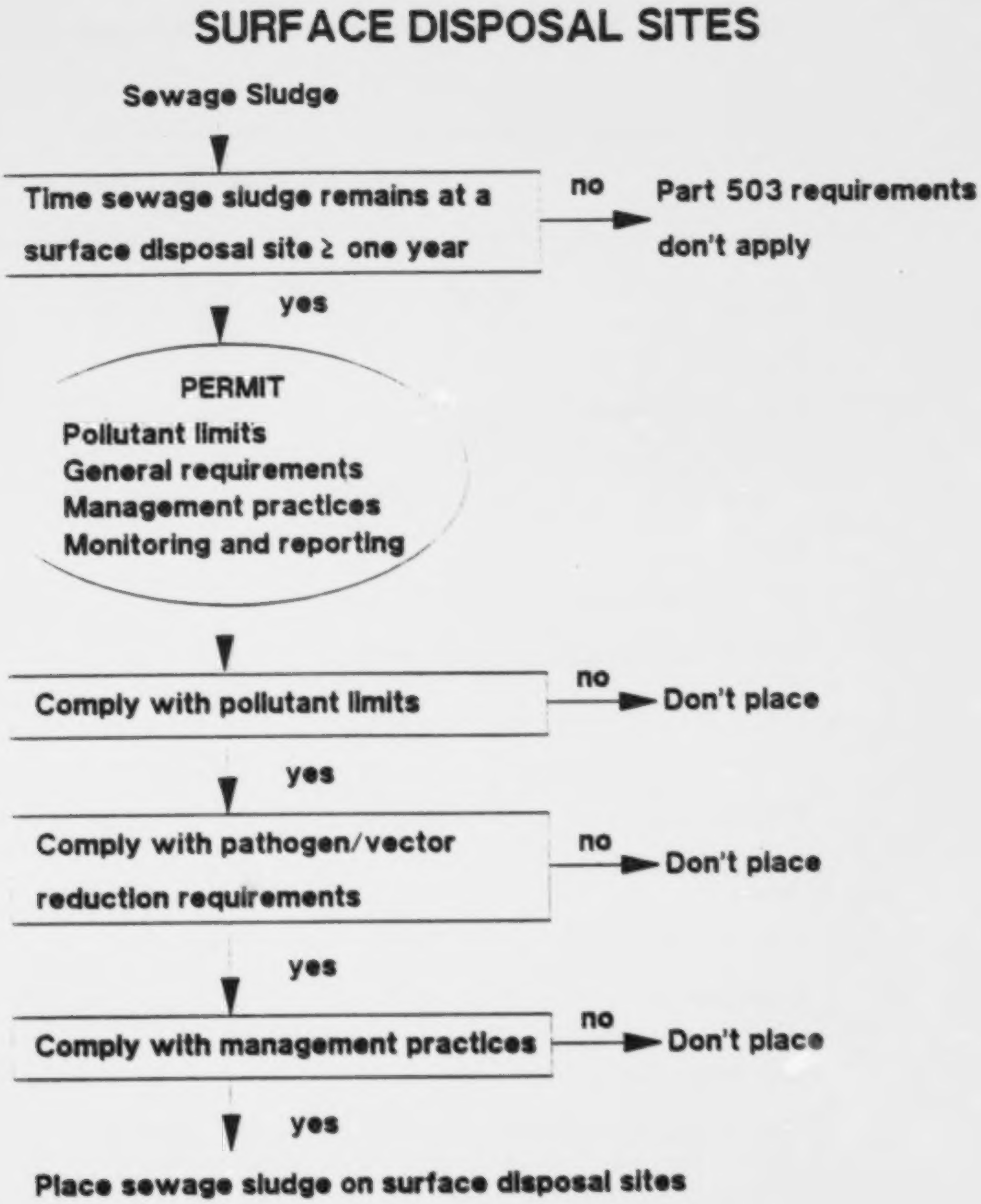
SURFACE DISPOSAL SITES
Sewage Sludge
Time sewage sludge remains at a surface disposal site ≥ one year
no
Part 503 requirements don't apply
yes
PERMIT
Pollutant limits
General requirements
Management practices
Monitoring and reporting
Comply with pollutant limits
no
Don't place
yes
Comply with pathogen/vector reduction requirements
no
Don't place
yes
Comply with management practices
no
Don't place
yes
Place sewage sludge on surface disposal sites

FIGURE 5-1. Surface Disposal Sites

SECTION SIX

GENERAL SITING REQUIREMENTS

When siting a monofill, special consideration must be given to evaluating sites near airports, 100-yr floodplains, wetlands, fault areas, seismic impact zones, and unstable areas. Requirements for siting monofills in each of these types of locations have been established by the proposed sewage sludge regulations to prevent adverse impacts to human health and the environment. The same rationale applies to the siting of surface disposal sites; therefore, the same requirements are made for both disposal practices. The following sections discuss the potential impacts associated with the six types of locations and present the rationales used to develop these requirements.

6.1 AIRPORT SITING REQUIREMENTS

An aircraft/bird strike is defined as any contact between a moving aircraft and a bird or group of birds. In response to the increasing problem of bird strikes at airports located near municipal solid waste disposal facilities, the Federal Aviation Administration (FAA) issued Order 5200.5 on October 16, 1974. The order states that landfills located within 10,000 ft of any runway used by turbojet aircraft, or within 5,000 ft of any runway used only by piston-type aircraft, should be closed. It also recommends the closing of any landfill located such that it places the airport runways or aircraft flight patterns between bird feeding, nesting, or roosting areas.

6.1.1 Impacts of Siting Facilities near Airports

The major problem in siting monofills near airports is the high incidence of bird collisions with airplanes. This hazard is caused by the large numbers of seagulls and other flocking birds that frequent landfill areas. The results of collisions between birds and aircraft range in severity from dents or holes in the aircraft structure to aircraft accidents and loss of life. Birds can cause aircraft accidents by penetrating windshields and disabling crew members or by striking and deforming the turbine engines or other aircraft parts (Solman, 1973). Bird strikes on windshields may crack, shatter, or completely destroy a panel, exposing pilots to flying pieces of glass, bird debris, and high wind velocity. In addition, bird strikes to windshields may startle pilots, impair their judgment, or distract their attention (Blokpoel, 1976).

The majority of bird strikes occur on the forward facing parts of the aircraft. Wide-bodied aircraft have a higher risk of colliding with birds because they have large frontal areas and large air intakes (Blokpoel, 1976). Table 6-1 presents statistics for military aviation in the United States in 1973, indicating the distribution of bird strikes by aircraft part struck (Blokpoel, 1976).

Because of high aircraft velocities, a bird strike can create an enormous impact. A 4-pound bird (such as a mallard duck) hit by an aircraft at 300 miles per hour (mph) causes an impact with a force of 15 tons over a saucer-sized area.

For a collision at 600 mph, the force of this impact would be nearly 60 tons (Solman, 1973). Statistics for civil aviation in the United Kingdom and the Soviet Union indicate that over two-thirds of all bird strikes occur at speeds of 92-184 mph (Blokpoel, 1976).

Bird strikes pose a greater hazard to jet aircraft than to propeller aircraft (Department of Interior, 1978). The rotating blades of a propeller aircraft tend to protect the piston engine, whereas turbine engines are usually exposed to direct bird ingestion, which can deform or destroy the rotor and stator blades of the engine's compressor (Blokpoel, 1976).

TABLE 6-1. Distribution of Bird Strikes by Aircraft Part Struck for U.S. Military Aviation (1973)

Aircraft Part	Percentage of Bird Strikes (%)
Wing	27.8
Tail	1.6
Nose	1.9
Windshield	5.5
Engine	32.8
Landing Gear	2.7
Other	17.7

Source: Blokpoel, 1976.

Observations of daily, seasonal, and operational patterns of bird strikes at airports have revealed that an estimated 75% of bird/aircraft collisions occur during takeoff and landing (Blokpoel, 1976). These collisions pose a serious hazard to aircraft because any sudden loss of power or control during this time can be particularly dangerous (Ultrasystems, Inc., 1977). Studies have determined that over 62% of bird strikes occur below altitudes of 500 ft (FAA, 1974). Air Force statistics for 1972 indicate that 86% of bird strikes occurred below 2,000 ft, and 93% occurred below 3,000 ft (Blokpoel, 1976). Impact altitude statistics compiled by the FAA (FAA, 1978) agree with these percentages.

6.1.2 Regulatory Requirement for Siting Monofills near Airports

Because each airport presents a unique topographical and ecological environment, uniform national standards for monofills located near airports are not appropriate. Each airport environment must be considered individually to assess the bird-strike hazard for that area. For this reason, EPA has established a broad, performance-based requirement for minimizing bird strike hazards posed by monofills. As for standards regulating solid waste disposal facilities (40 CFR Part 257) and those regulating municipal solid waste landfills (40 CFR Part 258), the requirements for sewage sludge monofills adopt the minimum distances from airport runways specified in FAA Order 5200.5. Sixty-two percent of all bird strikes occur below altitudes of 500 ft, and aircraft are generally below this altitude within the distances specified in the FAA Order (44 FR 53459, September 13, 1979). EPA therefore believes this restriction will prevent most bird/aircraft collisions caused by the proximity of monofills to airports, without placing an undue burden on the regulated community.

The distance from the runway to the boundary of the monofill is measured radially from the end of the runway to the boundary. The distance can be determined using recent United States Geological Survey (USGS) 7-1/2-minute maps, scale aerial photographs of the area, or plot plans of the facility and the surrounding area (EPA, 1980).

6.2 FLOODPLAIN SITING REQUIREMENTS

Large regions of the United States are susceptible to flooding (see Figure 6-1) (GCA, 1986a). Monofills sited in these regions can pose threats to human health and the environment in a variety of ways. Flooding can cause erosion of landfill cover material, allowing increased infiltration of water through the final cover or through any joints between covers and liners. Increased water content in the monofill can lead to increased leachate production, with increased potential for surface and groundwater contamination. The large water volumes and high flow velocity associated with floods can cause washout of wastes, also leading to surface-water and groundwater contamination. Wave action and flowing water can destroy containment structures and levees, and underseepage can weaken them. Weakened containment structures can increase the vulnerability of monofills during future flooding activity. Finally, flooding can relocate stream channels near the monofill, causing or worsening runon to the working face of the facility (MITRE, 1980; MRI, 1984).

6.2.1 Types of Floods Occurring in 100-yr Floodplains

The magnitude of a flood is usually described by its statistical frequency of occurrence. According to MRI (1984), "The differences between 10-, 100-, and 500-year floodplains are most commonly differences in elevation and size of area inundated." Floods occurring in floodplains can be divided into two categories: floods due to increased water elevation and coastal floods (MITRE, 1980).

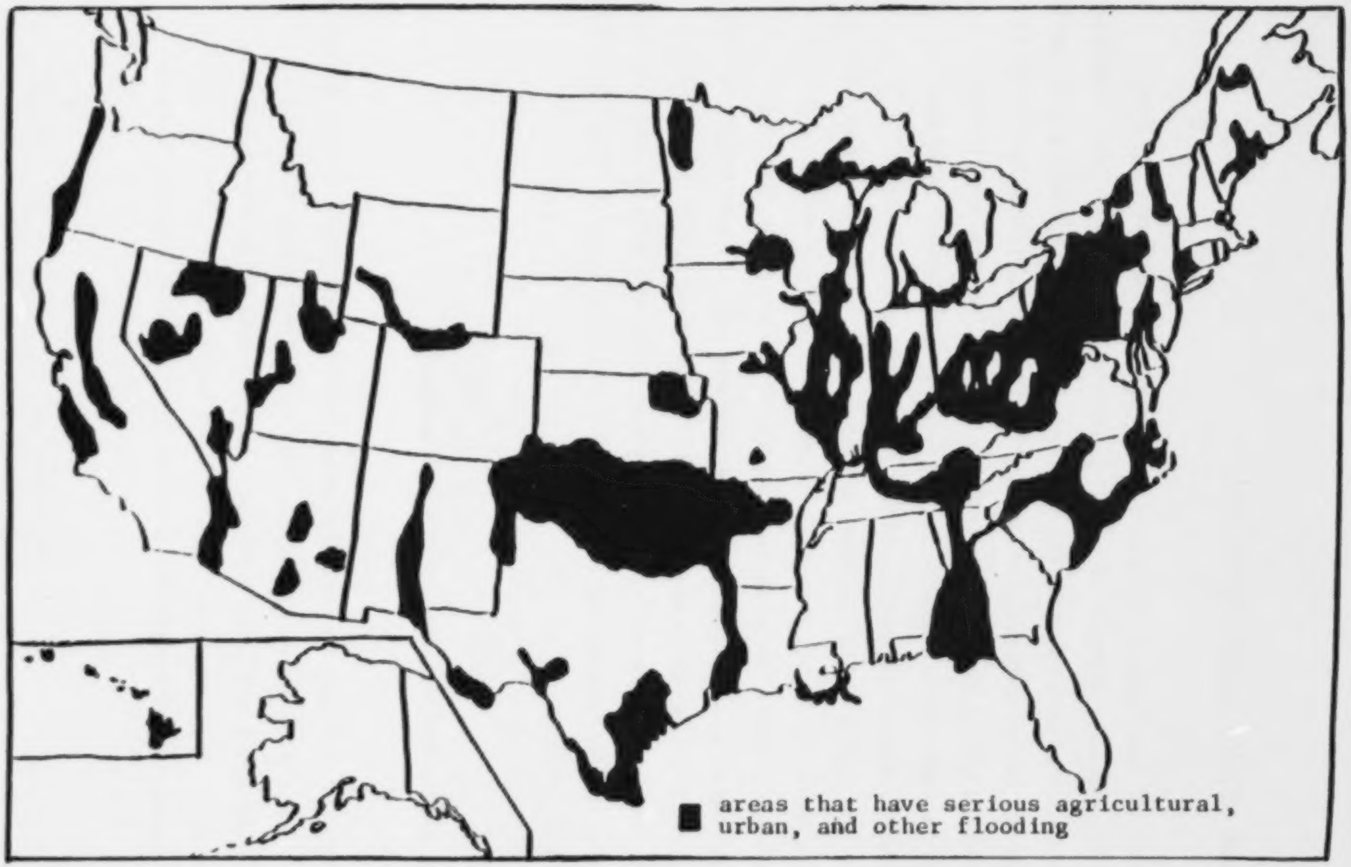
■ areas that have serious agricultural, urban, and other flooding

FIGURE 6-1. Existing Flooding Problems in the United States (Water Resources Council, 1978)

The types of flooding due to increased water elevation or flow velocity include riverine floods, shallow floods, sheet runoff, ponding, lacustrine floods, and alluvial fan flooding. These types are described below (MITRE, 1980):

- Riverine floods -- A riverine flood is the overflow of channelized water onto the surrounding floodplain. The velocity of a riverine flood can reach 30 ft/sec, which is enough to wash out the contents of a landfill.
- Shallow floods -- A shallow flood is an unconfined flood occurring in a topographically low area. Shallow floods include sheet runoff, which is "the broad unconfined downslope movement of water across gently sloping terrain" and ponding, which is "the accumulation of runoff or flow in a depression that has no natural or manmade subterranean or rim outlets, culverts, or pumping stations."
- Lacustrine floods -- A lacustrine flood results from the increase in elevation of a lake due to seasonal and long-term water fluctuations.
- Alluvial fan flooding -- Although flooding in an alluvial fan often follows the existing stream channels, unpredictable channel flow is common. Landfill containment structures can be threatened when the direction of channelized flow, predicted to be parallel to dikes and levees, becomes perpendicular. A flood with a velocity of 4 to 6 ft/sec can be sufficient to disintegrate an earthen levee.

Coastal flooding is defined as flooding accompanied by wave action. Facilities 500 ft inland can sustain severe damage during a 100-yr storm, and even facilities 1,000-2,000 ft inland have been known to sustain damage, particularly if they were not elevated above the height of the incoming waves (MITRE, 1980). Monofills in such locations would be subject to daily tides, wave setup, tsunamis (tidal waves), storm surges, and wind surges.

6.2.2 The History of the 100-yr Floodplain as an EPA Regulatory Standard

Executive Order (EO) 11988, signed into effect on May 24, 1977, requires that each Federal agency "shall provide leadership and shall take action to reduce the risk of flood loss, to minimize the impacts of floods on human safety, health and welfare, and to restore and preserve the natural and beneficial values served by floodplains in carrying out its responsibilities" (EO 11988, 1977). EPA addressed this order in the Criteria for Classification of Solid Waste Disposal Facilities (44 FR 53438, September 13, 1979) by requiring that "facilities or practices in 100-yr floodplains shall not restrict the flow of the base flood, reduce the temporary water storage capacity of the floodplain, or result in wash out of solid waste, so as to pose a hazard to human life, wildlife, or land or water resources" (40 CFR Part 257.3-1). EPA has used an almost identical requirement in its rule for municipal solid waste landfills and for sewage sludge monofills.

EPA considered many factors in its choice of the 100-yr floodplain as its regulatory standard. When EPA promulgated regulations for hazardous waste management facilities located in floodplains (40 CFR Part 264.18b), EPA considered using the 500-yr floodplain designation rather than the 100-yr designation. The U.S. Water Resources Council (WRC) recommends restricting "critical actions," defined as "any activity for which even a slight chance of flooding would be too great" in the 500-yr floodplain (43 FR 6030, February 10, 1978). Initially, EPA decided that hazardous waste management facilities were "critical activities" and proposed restricting their activity in the 500-yr floodplain. Because of almost unanimous opposition to the proposed standard (EPA, 1983), however, EPA adopted the 100-yr floodplain restriction in its final rule (46 FR 2847, January 12, 1981). This decision took into account the following (EPA, 1983):

- EO 11988 does not authorize regulation of activities in the 500-yr floodplain.
- EPA did not have sufficient information to justify use of the 500-yr floodplain designation.
- Maps of the 500-yr floodplain were available for only very limited portions of the United States. The development of 500-yr floodplain maps would place a large financial burden on the regulated community.
- Use of the 500-yr floodplain designation would restrict large areas of the United States, such as the mid-Atlantic region, the Southeast, and the Gulf States.

- The 100-yr floodplain is the designation established for use by the U.S. Geological Survey, the U.S. Army Corps of Engineers, the Federal Insurance Administration, and many other agencies.

The Agency determined, therefore, that the 100-yr floodplain was the appropriate regulatory standard for both hazardous waste facilities and sewage sludge monofills. The Agency's selection of the 100-yr floodplain standard was supported by analyses of present and past environmental impacts associated with washouts at waste management facilities. An examination of environmental impacts linked to the flooding of these facilities justified the regulatory view that the 100-yr floodplain criterion poses minimal environmental impact.

6.2.3 Impacts of Siting Facilities in Floodplains

The potential for contaminant releases from facilities located in flood-prone areas is considerable. The cause and severity of release can range from increased leachate production in inundated landfills to a complete washout of facilities. The destructive forces associated with floods depend on location, depth, velocities, flood duration, speed of onset, groundwater level, amount of debris in the flood waters, and chemical properties of the sewage sludge. These factors limit the time for implementation of emergency measures aimed at protecting the facility or removing wastes to a safe location. A discussion of these factors follows:

- Location -- Facilities located in the middle of a 100-yr floodplain are more likely to be inundated from all sides by the 100-yr flood, whereas facilities located on the edge of the 100-yr floodplain are more likely to have only one side exposed to the flood source (MITRE, 1980).
- Proximity of other structures -- Close proximity of other structures to landfills will increase floodwater elevations and can increase flow velocities downstream. Nearby structures can expose the landfill to channeling effects, and the landfill can cause or exacerbate a channel effect on structures downstream from the facility (MITRE, 1980).

- Flood depth -- Depth of the flood waters determines the strength of the hydrostatic forces in effect during the flood. Vertical loading can cause uplift and lateral displacement of wastes and is directly related to leachate generation. Horizontal loading can cause washout of wastes (MRI, 1984).
- Floodwater velocity -- Floodwater velocity affects the momentum and damage potential of the flood waters. As floodwater velocity increases, erosion of cover material tends to increase, causing damage to landfill structures. Floodwater velocity also affects the momentum and damage potential of the debris and sediment carried by the flood (MRI, 1984).
- Duration of flooding -- The duration of a flood affects the degree to which soils and building materials are saturated, contributing to their weakness, and increases the magnitude of leachate generation and leachate seeps (MRI, 1984).
- Speed of flood's onset -- Speed of a flood's onset is significant because it determines the warning time that monofill personnel will have to initiate emergency procedures to prevent washout of sewage sludge.
- Groundwater level -- Groundwater level at the time of flooding is an important factor in determining the impact of the flood on the integrity of landfill structures.
- Sediment and debris content of flood -- In addition to causing impact damage, sediment and debris from a flood can bury waste. Hidden from view, these wastes can potentially endanger human health and the environment.
- Chemical properties of sewage sludge -- the chemical characteristics of the sewage sludge in the monofill can affect their rate of dispersion following washout from the landfill. According to one theory (MRI, 1984), chemical constituents that tend to be widely diluted and dispersed by the flood waters cause the least adverse impacts on human health and the environment, whereas chemical constituents that remain concentrated in a small area will tend to cause greater impacts. Chemical properties that tend to increase the dispersion and dilution rates of constituents are water solubility, hydrolytic behavior and rate of volatilization, adsorption to suspended particulates and sediment, and biodegradation (MRI, 1984).

Because flooding is a surface-water phenomenon, surface waters are the major recipients of contaminants. When contaminants enter surface waters, they migrate rapidly to downstream waterways. The rapid transport processes produce large areas of surface water contamination. In coastal or estuarine areas, tidal effects can disperse contaminants both upstream and downstream of the source. Although mixing and dispersion generally dilute contaminant concentrations far from the source, contaminant adsorption on sediments can produce a long-term problem. Contaminated sediments will be both dispersed over the inundated land surfaces and transported downstream as part of the river sediment load after the flood subsides, resulting in widespread contamination.

Many contaminants that tend to adsorb on sediments (i.e., many organics and heavy metals) will remain long after soluble and suspended contaminants have been swept downstream. Suspended contaminants will tend to settle in slow-moving portions of the river or downstream reservoirs. In coastal areas, they may spread up and down the coast as well as offshore, producing long-term contamination problems that can adversely affect downstream water supplies and aquatic resources many years after the source of contamination has been cleaned up. Contaminants in sediments will continue to leach back into surrounding surface waters and contaminate surface waters downstream. There, they may destroy benthic organisms or enter the benthic food chain, resulting in toxic effects at higher trophic levels, i.e., important fisheries. Recreational use of the waters may also be impaired.

Contamination of surface waters and sediments due to releases from waste management facilities can destroy aquatic organisms directly through toxic effects, including accumulation of low-level releases in the food web, and indirectly through destruction of habitat. Contaminated sediments may destroy spawning grounds for fish and kill shellfish and other benthic invertebrates that inhabit the sediments. Fisheries may be damaged in turn due to the elimination of the benthic food supply or through the consumption of contaminated organisms by fish. Destruction of spawning habitat and toxicity of the waters to sensitive, early life stages may also limit the reproductive success of fisheries. Other indirect effects include changes in species composition and diversity when sensitive species are eliminated and replaced by less desirable pollution-tolerant species. In severe cases, all organisms near the release may be destroyed.

The release of wastes to surface waters also may result in both acute and chronic human-health effects. The primary route of exposure is through the ingestion of drinking water. The ingestion of fish and other aquatic organisms that accumulate chemicals is also of concern. Human-health effects may exist at lower levels of bioaccumulation than those which affect indigenous aquatic organisms. Recreational use of surface water for water-contact activities such as swimming can also result in contaminant exposure. A major concern associated with the effects of surface-water contamination on human health is that existing ambient water-quality criteria for human toxic and carcinogenic protection are in many cases below the detection limits in water. Potentially toxic releases from waste management facilities can therefore go undetected. Additionally, the human-health effects of chronic exposures may not be exhibited for decades.

Surface waters represent an economic asset to regional and local areas. Contaminant releases to surface waters that result in widespread, long-term degradation of water quality can affect agricultural and industrial use, i.e., food processing. Any degradation of water quality will result in a loss to the community.

6.2.4 Preventive Measures and Emergency Responses to Flooding of Facilities

It is difficult and expensive to design preventive measures that can withstand the forces of a major flood. The two major approaches are (1) flood protection measures that rely on diversion structures such as levees, dikes, and floodwalls to divert the flood waters away from units and prevent inundation and (2) flood-proofing measures that allow the flood waters to come in contact with units, but that include design features to prevent damage. The latter approach includes designing fencing units to prevent impacts from floating debris, grading landfills to maximize runoff and minimize erosion, and providing drainage systems to facilitate the collection and removal of runon and runoff (MITRE, 1980).

Emergency responses to contaminant releases caused by flooding are difficult to implement and generally not very effective. In the event of a washout or major release, wastes usually cannot be recovered. Preventive measures, such as moving waste to a safe location or erecting flood protection structures prior to an encroaching flood, are also difficult to implement due to the difficult working conditions and the general disruption of transportation systems (such as access roads) and other services that generally occurs during floods.

When contaminants are released into surface waters, they spread rapidly over large areas. This rapid spreading may result in long-term sediment contamination that extends far downstream, as well as soil contamination in downstream land areas inundated during the flood. These problems may persist for many years after the facility has been cleaned up. Offsite removal of contaminated sediments generally is not feasible due to the large extent of the affected areas.

It is generally impossible to monitor a release during a flood because rapid transport and dilution typically move the peak concentrations of surface water contaminants far downstream before they can be measured. Site access is also a problem during a flood. Even if the site is accessible, a major sampling effort would be required to characterize the temporal and spatial variations in pollutant concentrations. These variations are very important because of the dynamics of the contaminant release and flow conditions. Because of the difficulty in assembling a sampling team given the short lead time and hazardous working conditions during floods, the monitoring of sediments and surface-water concentrations usually must wait until after the flood has subsided. Monitoring fish-tissue concentrations and biological impacts such as changes in abundance, species composition, and diversity also is important, but this task involves a major long-term effort because important changes often occur gradually.

6.2.5 Case Studies

EPA has examined case studies that depict many of the problems associated with locating waste management facilities in floodplains (EPA, 1988a). These studies illustrate (1) the potential for waste releases in floodplains, (2) the common release pathways, and (3) the potential impacts associated with these releases. The observed waste releases were caused by flooding of waste sites, dike failures, streambank erosion, contaminated runoff, or shallow groundwater contamination recharging a nearby stream. In all cases, continuous or frequent release to nearby surface waters occurred due to failures in monofill design and operating specifications. Flooding, however, had been responsible for rapid dispersion of a large quantity of contaminants. Environmental impacts observed included surface-water contamination, fish and aquatic organism kills, and long-term sediment contamination.

6.2.6 Regulatory Requirement for Siting Monofills in Floodplains

EPA's requirement states that a sewage sludge monofill shall not restrict the flow of a base (100-yr) flood; reduce the temporary water storage capacity of a floodplain; or present a hazard to human health, wildlife, and land and water resources because of a washout of sewage sludge. Monofills can employ a variety of flood controls to meet this requirement, several of which follow (MITRE, 1980):

- Slope -- Location of monofills below grade causes ponding that can carry wastes out of the monofill in solution or suspension and increase leachate production. A unit slope greater than 2% but less than 5% will inhibit the ponding effect, but will not lead to increased erosion.
- Drainage capacity -- A leachate collection system can be used during emergencies to increase the drainage capacity of a landfill. Permanent drainage channels or ditches should be constructed on the uphill side of the monofill to divert water around the site. Portable drainage structures can be used during flooding; however, the speed of flood onset must be considered in choosing such structures for flood protection.

- Fencing -- Any fencing used around a monofill should be permeable. Permeable fences will allow water to pass through, making them less likely to be carried away by flood waters and reducing the potential impact damage caused by debris in the flood waters.

- Diversion structures -- Levees are the most commonly used diversion structure; however, flood walls and ditches can also be used. Levees are generally constructed of earthen materials and are designed to provide flood protection from seasonal high water. They are, however, subject to failure from underseepage as a result of flood inundation. Flood walls are constructed of construction materials such as concrete or steel sheet piping and are designed to prevent inundation of adjacent land.

The first step in determining compliance with the requirement is to determine whether the monofill is located in a 100-yr floodplain. This information can be obtained from existing permits and operation applications or from 100-yr floodplain maps. These maps are available from the following sources (EPA, 1980):

- State flood control agencies or other departments
- Federal Emergency Management Agency (FEMA), Flood Insurance Rate Map (FIRM), or Flood Hazard Boundary Map (FHBM)
- Local and regional planning and zoning agencies
- Soil Conservation Service, U.S. Department of Agriculture
- U.S. Army Corps of Engineers
- National Oceanic and Atmospheric Administration
- Federal Housing Administration (HUD)
- U.S. Geological Survey
- Bureau of Land Management, U.S. Department of the Interior
- Bureau of Reclamation, U.S. Department of the Interior
- Tennessee Valley Authority
- River basin commissions and special flood control districts
- Local and State agencies involved with public works construction (i.e., bridges, culverts, highways, channel improvements, and urbanization studies)

Additional assistance can be obtained from the State agency charged with flood protection or floodplain management, any of the map-source agencies with the necessary expertise previously listed, or a qualified professional firm.

After determining location with respect to 100-yr floodplains, an owner or operator should determine if the facility is protected from washout by a 100-yr flood (EPA, 1980).

Types and effectiveness of washout protection used in each area of the facility below the 100-yr flood level should be considered. Washout protection may include:

- Dikes
- Levees
- Berms
- Flexible linings
- Vegetative cover
- Riprap
- Diversion of high velocity flows around the facility
- Change in soil matrix by chemical alteration

Flood flow velocity should also be considered (EPA, 1980). A flood flow velocity of at least 2.5 times the average velocity over the entire floodplain cross section should be used for those portions of the facility in the floodway. A minimum value of 1 times the average velocity over the entire floodplain cross section should be used for those portions of the facility in the floodway fringe. That is,

$$\bar{v} = Q/A$$

where $\bar{v}$ - average velocity

Q - 100-year flood flow (cfs)

A - cross-sectional area of floodplain

Matrices comparing the efficiency of each type of washout protection with different flood-flow velocities are available in the U.S. Department of Transportation's Hydraulic Engineering Circular No. 15 and the U.S. Army Corps of Engineer's Shore Protection Manual (EPA, 1980). These matrices can be used to help evaluate the adequacy of protection for each facility.

6.3 WETLANDS SITING REQUIREMENTS

The wetlands of the United States, composed in part of inland and coastal swamps, bogs, marshes, sloughs, mudflats, wet meadows, natural ponds, and river overflows, serve as important natural hydrologic, ecologic, social, and economic resources. Scientific studies have come to recognize wetland habitats as essential breeding, rearing, and feeding grounds for numerous species of fish and wildlife, including many endangered species. These ecologically sensitive habitats have been found to play important roles in both flood protection and pollution control. The general public, government, and business, however, have had little concern for wetlands protection, thus these areas have been inconsistently regulated by Federal, State, and local governments. Such policies resulted in the direct destruction of an estimated 48 million acres of wetlands through draining, dredging, and landfilling for agricultural, residential, and commercial development (43 FR 4942, February 6, 1978). A recent study by the Office of Technology Assessment (OTA, 1984) concluded that, based, on current policies and trends, an estimated 300,000 acres of wetland resources are lost each year. Increasing scientific understanding of the functions and values of wetlands indicates that this conversion will cause serious economic consequences throughout the country (Baldwin, 1985).

Despite recent concern by the public and State and Federal regulating authorities, wetland protection programs are neither comprehensive nor consistent (OTA, 1984). The Clean Water Act (CWA) (Pub. L. 92-500 as amended by Pub. L. 92-217 and Pub. L. 95-576; 33 U.S.C. 1251 et seq.) stands alone in addressing adverse effects to wetlands by regulating impacts to "waters of the United States." An expansive definition of the latter has been viewed to encompass wetlands indirectly. The ambiguousness of the statutory language however, has lead to disputes over which Federal agency or agencies have primary authority to implement a Federal regulatory program to protect wetlands from waste disposal. The result has been an inconsistent and often incomprehensive wetland protection system, ultimately mandating a reevaluation of present policy and issuance of a natural wetlands protection standard administered solely through and by the EPA.

The term "wetlands" refers to those areas inundated or saturated by surface or ground water at a frequency and over a duration sufficient to support a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands include, but are not limited to, swamps, marshes, and bogs. The justification for protecting wetlands focuses on the important ecological functions and resource values that wetlands provide, including the following (OTA, 1984):

- Wetlands store runoff and slow the downstream flow of water, thus reducing floodpeaks and frequency of flooding in downstream areas.
- By temporarily or permanently retaining pollutants, wetlands can improve, to varying degrees, the quality of the water that flows over and through them.
- Wetlands provide food and habitat for many game and nongame animals. Approximately 20% of all plant and animal species listed by the Federal government as threatened or endangered depend heavily on wetlands.
- Wetlands may significantly reduce shoreline erosion caused by large waves and major coastal and riverine flooding.

- Some wetlands are recharge areas to groundwater systems that supplement local or regional groundwater supplies through infiltration/percolation of surface water.

6.3.1 Impacts of Siting Facilities in Wetlands

Wetland environments may be adversely affected in several ways by the proximity of solid waste land disposal facilities. Depositing wastes into wetlands causes physical disturbances and creates potential for the accidental discharges of wastes into the ecosystems (GCA, 1986b). These physical and/or accidental impacts can directly affect the chemical, biological, and human-use characteristics of the aquatic ecosystems. The rule 40 CFR 230.42(a)(3)(b) states:

> The loss of wetlands....is likely to damage or destroy habitat and adversely affect the biological productivity of wetland ecosystems by smothering, by dewatering, by permanently flooding, or by altering substrate elevation or periodicity of water movement. The addition of materials may destroy wetland vegetation or result in advancement of succession to dry land species. It may reduce or eliminate nutrient exchange by a reduction of the systems productivity, or by altering current patterns and velocities. Disruption or elimination of the wetlands system can degrade quality by obstructing circulation patterns that flush large expanses of wetland systems, by interfering with the filtration function of wetlands, or by changing the aquifer recharge capability of a wetland. Accidental discharges can change the wetland habitat value for fish and wildlife. When disruptions in flow and circulation patterns occur, apparently minor loss of wetland acreage may result in major losses through secondary impacts.

The degradation of wetlands by foreign substance intrusion can cause the destruction of recreational and commercial fisheries. Placing waste materials in wetlands can affect the suitability of recreational and commercial fishing grounds as habitat for consumable aquatic and terrestrial organisms by destroying migration and spawning areas, as well as food sources (40 CFR Part 230.51b). These natural aquatic areas also provide aesthetic, recreational, scientific, and educational values to the public.

In addition, wetlands are commonly near to or contain surface waters and by definition have saturated soils and a shallow water table. Therefore, the release of wastes often affects surface waters and ground waters surrounding the wetlands. Surface water transport results in extensive surface water and sediment contamination. The difficulties in performing corrective actions for widespread contamination and the extensive wetland disruption involved in sediment removal have been documented in case studies (EPA, 1988a).

6.3.2 Current Federal Protection Measures

President Carter advanced Federal protection of wetlands in 1977 by signing EO 11990 for protection of wetlands and EO 11988 for floodplain management (EO 11988 is discussed in Section 6.2). These Federal directives prescribe measures to be taken by Federal agencies to avoid adverse impacts on wetland and floodplain resources (GCA, 1986b).

EO 11990 articulates a strong Federal policy favoring the protection of wetlands. It directs Federal agencies, in carrying out their responsibilities, to minimize the destruction or degradation of wetlands and to preserve and enhance their natural and beneficial values. It also requires Federal agencies to avoid undertaking or providing assistance for new construction, including draining, dredging, and filling, in wetland areas unless there is no practical alternative and only if measures are taken to minimize harm to wetlands. In principal, EO 11990 appears to control Federal activities involving waste disposal in wetlands. In practice, however, because Federal directives like this one are largely enforced by the Federal agencies themselves, it is difficult to measure the extent to which Federal waste disposal in wetlands has been affected. In addition, this Executive

Order does not apply to private, non-Federal waste disposal activities in wetlands (GCA, 1986b).

The ineffectiveness of the present Federal system is demonstrated by EPA's use of its veto authority under §404(c) of the CWA. Although the U.S. Army Corps of Engineers releases about 11,000 permit requests each year, permit denials are estimated to be less than 3%, with only a few §404(c) veto actions being taken by EPA. Only about a dozen permit controversies with the Corps have been evaluated by regional offices for resolution to the Washington headquarters under §404(c) procedures. The rare use of §404(c) reflects the unusual role created by Congress for EPA as overseer of another agency's regulatory program (Baldwin, 1985).

Estimates of current national wetlands loss also exemplify the problems with Federal implementation. Approximately 99 million acres of wetlands are located in the 48 contiguous States, comprising 5% of the total area (EPA, 1987b). Of these 99 million acres, between 300,000 and 500,000 acres of wetland resources are adversely affected annually. Fifty-thousand acres of this wetlands loss are permitted under §404. An unknown amount of loss, occurring through discharges that could come under §404 jurisdiction but that are currently unregulated or illegal, is probably equal to at least the amount of permitted loss (EPA, 1986). EPA believes these unnecessary losses can be prevented through a comprehensive and consolidated Federal system that does not depend on the interactions of different agencies with different Federal (EPA) administrative and substantive duties. Such a policy would have as its ultimate goal the development of national criteria and standards.

6.3.3 Regulatory Requirement for Siting Monofills in Wetlands

The sewage sludge monofill criteria include provisions that impose a ban on locating monofills in wetlands. EPA believes this ban is necessary because monofills cause irreparable harm to these sensitive ecosystems. This decision is consistent with Agency policy for the protection of the Nation's wetland resources. This policy of wetlands protection has been articulated over the years in various Agency directives on wetlands, beginning with the policy statement issued by the first EPA Administrator in 1973. It also is embodied in regulatory form in the guidelines for specification of Disposal Sites for Dredged or Fill Material (40 CFR Part 230) effective under §404(b)(1) of the CWA.

Recently, the Agency has identified wetlands protection as a top priority, aggressively implementing the §404 program, increasing enforcement against illegal discharges, and specifying other measures as necessary. To this end, the Agency considers the wetland requirement an essential measure for protecting wetlands resources.

For consistency with the CWA, the revised criteria adopts the definition of wetlands used in the §404(b)(1) guidelines. These guidelines, first promulgated in 1975 and then amended in 1980, specify the analytical tools to be used in evaluating and testing the impact of dredged or fill material discharges on waters of the United States, including wetlands. Fundamental to the guidelines is the precept that discharges into aquatic ecosystems should not be allowed unless it can be demonstrated that such discharges will not have an unacceptable effect. In particular, the guidelines identify filling operations in wetlands as among the most severe environmental impacts. For this reason, the guidelines are directly relevant to solid waste disposal in wetlands and provide the basis for decision making with respect to such activities.

6.4 FAULT AREA SITING REQUIREMENTS

Earthquakes are part of the Earth's natural dynamics. When they occur, they are usually catastrophic, causing the loss of life and property. North Americans typically associate the west coast of the continent with areas of known seismic activity, but serious earthquake shocks have also occurred in and near the east coast (Legget and Karrow, 1983).

Earthquakes are usually caused by movements along faults, which are fractures in rock along which the adjacent rock surfaces are differentially displaced. Faults may vary in length from a few meters to many kilometers. The presence of faults indicates that at some time in the past movement has occurred by either a slow slip (movement along the fault plane) that produces no ground motion, or a sudden rupture, which results in the perceptible ground motion known as an earthquake (Bolt, 1978).

Faults assume different geometric forms: normal, reverse and strike-slip. A normal fault has movement down the dip of the rock, while a reverse or thrust fault has movement up the dip of the rock. Since vertical displacements seen at the surface are produced by both normal and reverse faults, both of these faults are referred to as dip-slip faults. A strike-slip fault has a relative displacement at a right angle to the dip, as viewed from the ground surface. This type of fault is sometimes referred to as a wrench or tear fault (Legget and Karrow, 1983). Strike-slip or dip-slip faults can cause the partial or total collapse of buildings, bridges, and tunnels; failure of manmade and natural slopes; and damage to water or other types of distribution systems.

When displacement along a fault occurs, movements of great masses of material naturally develop dynamic effects of magnitude that initiate vibrations or seismic waves within the earth's crust. The seismic waves travel for great distances in all directions, thereby creating effects over a wide area (Legget and Karrow, 1983). Section 6.5 discusses seismic zones in more detail.

The fracturing that faults represent is a normal part of the dynamic process of geologic development. Therefore, some faults are of great age, whereas others seen in Pleistocene deposits are, correspondingly, very young geologically. The age of a fault is often related to whether or not it is active; young faults tend to be more active. Only in active faults is movement on the fault line expected to take place in the foreseeable future.

The faintest possibility of movement on a fault plane under any engineering structures, however, should be avoided, even to the extent of abandoning a proposed building site (Legget and Karrow, 1983).

Damage can also be caused by the displacement of the fault itself. This particular earthquake hazard is very limited in area and can usually be avoided by obtaining geologic advice on the location of active faults before undertaking construction (Bolt, 1978).

6.4.1 Impacts of Siting Facilities in Fault Areas

Constructing a monofill over an active fault can have serious consequences. Facilities located directly over a fault may be damaged, for example, by lining rupture, during the fault displacement. Lining rupture may result in an uncontrolled release of contaminants into the environment.

Investigating sites for surface-fault rupture is a deceptively difficult geologic task. Many active faults are complex, consisting of multiple breaks, yet the evidence for identifying active fault traces is generally subtle or obscure. The distinction between active and long-inactive faults may be difficult to make. To assist with this task, the California Division of Mines and Geology has developed guidelines for detecting and evaluating the hazard of surface and near-surface fault rupture (California, 1975).

The general fault zone can be divided into a main fault zone, a branch fault zone, and a secondary fault zone. The main fault zone contains the main fault, i.e., the fault with the greatest displacement or length, and closely associated faults. The width of this zone ranges up to 3,000 ft, but in most cases is half that width or less. Faults that diverge from and extend well beyond the main zone of faults are referred to as branch faults. Secondary faults are completely separate from the main fault, sometimes several hundred feet to a few miles away. Often associated with main, branch, and secondary faults are small, subsurface faults that are evident as fault planes running parallel to the fault and typically are considered a part of that fault.

Adjacent to the fault rupture a zone of deformation is commonly found. This is an area where the ground has been bent or warped as a consequence of the two surface planes moving against one another. Surface deformation may occur within a zone several tens to several hundred feet wide. Structures located within this zone are subject to distortion and are likely to be damaged.

6.4.2 Regulatory Requirement for Siting Monofills in Fault Areas

The Agency believes that monofills may be affected by fault rupture and surface deformation resulting from earthquakes. To minimize the risk, the rule states that new sewage sludge units shall be located 60 m (200 ft) or more from a fault displacement from Holocene time. (The Holocene is a geologic time period, known as an epoch, that extends from the end of the Pleistocene to the present and includes approximately the last 11,000 yr). The proposed regulation should prevent siting on the zone of deformation where the ground may be bent or warped.

The Agency's proposal reduces the possibility that landfill slopes would be impaired by fault rupture, and the resulting potential for exposing waste to surface runoff.

6.5 SEISMIC IMPACT ZONE

Another aspect of the effect of earthquakes on monofill siting is the extent of the seismic impact zone. Geologic features can contribute significantly to the local effects of earthquake shock. Because seismic waves travel at different speeds in different materials, the effects of seismic waves in rock and unconsolidated material will vary (Legget and Karrow, 1983).

Larger areas are influenced in the eastern United States than in the western United States (USGS, 1981).

Regional seismicity maps have been available since the 1950s to aid the engineer in reducing earthquake effects on buildings and other structures. Regional seismicity or risk maps usually do not attempt to reflect geologic conditions or take into account variations due to soil properties. The maps primarily provide insight into the relative hazard of seismicity across the United States together with the relative importance of various parameters involved (Algermission and Perkins, 1976; and Bolt, 1975).

The U.S. Geologic Survey published an open-file report addressing the probabilistic estimate of maximum acceleration in rock in the contiguous United States. The report presents maps of the relative earthquake hazard in various "seismic zones" based on a constant-probability level. Maps provided with the report identify the maximum horizontal accelerations in hard rock expressed as a percentage of the Earth's gravitational pull, g, with a 90% probability that they will not be exceeded in 10, 50, and 250 years (Algermission and Perkins, 1982).

A single parameter, such as an estimate of maximum acceleration, does not provide all of the information necessary to describe the characteristics of strong ground motion important in structural design. Nevertheless, a wide range of structures can and have been designed to be earthquake resistant using peak acceleration as the basic ground motion data (Algermission and Perkins, 1976).

6.5.1 Impacts of Siting Facilities in Seismic Impact Areas

Earthquakes can affect a waste management facility through ground motion, surface faulting, earthquake-induced ground failures, and tsunamis. Although earthquakes cause much less economic loss annually in the United States than do ground failures and floods, major earthquakes have the potential to cause sudden and great loss. Ground motion associated with earthquakes can damage a monofill by causing dikes and berms to fail, resulting in the exposure of waste and/or rupture of any leachate collection and liner systems. Runoff contaminated by contact with exposed waste potentially could contaminate surface water bodies.

Failure of natural and manmade slopes adjacent to the monofill may also affect the facility. Failure of these slopes may damage runon and runoff control systems, leachate disposal/management systems, and the stability of the containment structure berms. These earthquake effects constitute one important set of considerations in siting, designing, and constructing these critical structures (EPA, 1988a).

Earthquakes that occur near waste management facilities pose a risk to the public and the environment because the motion of the ground during destructive earthquakes has the potential to damage inplant collection and waste-processing facilities, i.e., landfills with protective engineering design and long-term surveillance. The most typical seismic-related damage categories are:

- Failure of structures, tanks, etc., due to a high level of ground motion
- Failure of structures, tanks, impoundments, and landfill containments due to soil liquefaction and liquefaction-induced landsliding and/or lurching, or, generally, extensive soil settlement, failure of structures and tanks due to fault rupture passing through structures, tanks, and landfills
- General landsliding
- Failure of impoundments (dams, embankments, etc.)

One major cause of destruction during an earthquake is the failure of the ground structure by loss of strength. This type of failure typically occurs in loose, saturated, cohesionless soils. This phenomenon, termed liquefaction, is a result of an increase in the soil pore water pressure, that decreases, and sometimes eliminates, the shear strength of the soil.

Bodies of loose, relatively fine-grained uniform sand below the water table are susceptible to liquefaction during an earthquake, especially if the duration of the quake is long enough for the occurrence of a large number of oscillations involving repeated reversals of shearing strains of large amplitude. Soil that has lost all strength behaves like a viscous fluid and often appears in the form of "sand fountains or boils" during earthquakes. When a soil fails in this manner, a structure resting on it simply sinks (Prakash, 1981; Terzahgi and Peck, 1967).

Liquefaction causes three types of ground failure: lateral spreads, flow failures, and loss of bearing strength. Lateral spreads involve the lateral movement of large blocks of soil as a result of liquefaction in a subsurface layer. Flow failures consist of liquefied soil or blocks of intact material riding on a layer of soil, and are the most catastrophic type of ground failure caused by liquefaction. Loss of bearing strength when the soil supporting a building or structure liquefies causes large deformations within the soil that allow the structure to settle and tip (USGS, 1981).

6.5.2 Regulatory Requirement for Siting Monofills in Seismic Impact Areas

The Agency believes that monofills may be affected by ground motion resulting from an earthquake. To minimize the risk, all new monofill units located within a "seismic zone" defined as having a 10% probability of maximum horizontal acceleration in hard rock exceeding 0.10 g in 250 years must design all containment structures, including liners, leachate collection systems, and surface water control systems, to resist the maximum horizontal acceleration for the zone.

This requirement also applies to a seismic zone with a 4% probability of exceeding the maximum horizontal acceleration in 100 years. The Agency believes that the areas affected by the seismic zone requirement represent the areas of the United States with the greatest seismic risk. In addition, the

Agency believes that these probabilities and ranges of maximum horizontal accelerations are acceptable, and that the requirement adequately protects human health and the environment.

EPA has included this restriction as a performance requirement for monofills to minimize the risk of slope and liner failure due to seismic activity. By minimizing the risk failure of the landfill slopes, the requirement also reduces the potential for exposure of sewage sludge to the atmosphere and the possible contamination of runoff by contacting exposed sewage sludge.

In addition, the Agency requires that no new monofill units shall be located in areas where support for the structural components is inadequate. The Agency considers soils with the potential for liquefaction as areas of inadequate support. An owner or operator must determine if the proposed monofill site is susceptible to liquefaction. If it is, and protective measures cannot be incorporated into the design, then the Agency will consider the site unstable. Unstable areas are further discussed in Section 6.6 of this document.

6.5.3 Methods for Determining Seismic Risk

Numerous seismic design methods are available to the owner or operator to assess the proposed design of the monofill and appurtenances. The two most commonly used include the seismic coefficient method and the permanent displacement method. Both methods have been used to evaluate the design of embankments under seismic loading conditions.

The seismic coefficient method utilizes a static, horizontal inertial force applied to a potential sliding mass in an otherwise conventional static limit analysis. This method, commonly referred to as a conventional static stability analysis, represents earthquake loading by a statically applied horizontal force. The seismic force is proportional to the weight of the potential sliding mass multiplied by the seismic coefficient, which is some fraction of the acceleration of gravity (Lambe and Whitman, 1969; Hynes-Griffin and Franklin, 1984).

The permanent displacement analysis estimates permanent displacement in an embankment due to seismic loading, as discussed by Newmark in his 1965 Rankin Lectures. A principle parameter in the analysis is the peak bedrock acceleration determined by a site-specific risk assessment. Using this method, the motions of a system consisting of a rigid block sliding on an inclined plane are analyzed. If the base inclined plane is subjected to some sequence of acceleration pulses (the peak bedrock acceleration) large enough to induce sliding of the block, the block will come to rest at some displaced position down the slope. An embankment response analysis of the containment structure must then be performed. This dynamic analysis requires shear wave velocities and damping values, consistent with expected strain levels, for all materials comprising the structure (Hynes-Griffen and Franklin, 1984).

The Department of the Army Waterways Experiment Station published a miscellaneous paper outlining the procedures for performing a permanent displacement analysis. The paper presents figures and tables to assist in the analysis, as well as to indicate situations where the procedure is not applicable (Hynes-Griffen and Franklin, 1984).

6.6 SITING REQUIREMENTS FOR UNSTABLE AREAS

EPA has chosen to ban the siting of monofills in areas that are determined to be geologically unstable. These include landslide-prone regions, areas with comprehensive or expansive soils or ultra-sensitive (quick) clays, and subsidence-prone areas, including karst terrains.

6.6.1 Types of Unstable Areas

6.6.1.1 Landslide Prone Areas

Landslide is a general term covering a wide variety of mass-movement land forms and processes involving the downslope transport of soil and rock material under gravitational influence. Landslides are a significant hazard in virtually every State, and often take place in conjunction with other hazards such as earthquakes, floods, and volcanoes (USGS, 1981). They occur when earth materials fail under shear stress, and can be triggered by any human activity or natural event that increases shear stress or lowers shear strength. The major causes of landslides are (Winterkorn and Hsai-Yang, 1975):

- Construction operations or erosion
- Earthquakes and vibrations
- Rains or melting snow
- Freezing and thawing
- Dry spells
- Seepage from manmade sources of water

Construction operations or erosion can cause a landslide if they affect the support of a slope. Slides are common in excavated cuts for highways and railways and also in quarries and pits. Similarly, erosion of the toe (bottom) of a slope can leave the remaining slope face unsupported and subject to sliding. Heavy buildings located close to the edge of a slope can also initiate a slide.

Earthquakes and vibrations from blasting or construction-related operations can cause spontaneous liquefaction of loose sand, silt, or loess deposits situated below the groundwater table. Under similar circumstances, some sensitive clays can undergo a decrease in shear strength. If pore pressures increase to the point of total overburden pressure, the shear strength of the soil is drastically reduced, causing the soil to flow downhill like a heavy liquid.

Extremes of temperature and precipitation can contribute to landslides. Rains or melting snows can cause increased pore water pressures leading to reduced shear strengths along potential slip faces. Most landslides occur after heavy rains or during spring snow melts when large quantities of water can penetrate cracks and fissures. Freezing or thawing can induce cracking in rock formations that result in rockslides. In silty soils, the freeze-thaw cycle can increase pore pressures and, consequently, ground surface movements. Drying also may result in crack formations that reduce the soil's shear strength, thereby increasing the chance of slides.

6.6.1.2 Expansive Soils

Certain types of soils and soft rocks will expand when they become wet and shrink when they dry out. Expansive soils are generally rich in clay minerals, which swell by adsorbing water that enters and expands the spaces between the mineral's crystalline structures. Upon drying, the spacing decreases and the clay shrinks. The amount of expansion that can occur depends on the type and quantity of clay mineral present, and is a function of time, weight of material resting on top of the expansive clay (confining load), initial density, and initial water content (Department of Interior, 1974). Montmorillonite clays are the most prone to swelling with bentonite clay an extreme case that can increase in volume by a factor of 10 from its dry to its saturated state.

Expansive soils are widely distributed worldwide. Montmorillonites are most abundant in geological formations throughout the Rocky Mountains, in the upper Great Plains, in the southern gulf coast plain, and along the Pacific coast. They are also locally abundant throughout the Great Basin region, and along the Atlantic coast.

The extent of damage causd by expansive soils to houses, commercial buildings, streets, buried utilities, and other structures was estimated at $7 billion in 1980. Expansive clay formations that do not exhibit natural excessive shrink-swell activity can become a problem as a result of construction activities that reduce confining load or increase water access.

6.6.1.3 Subsidence Areas

Subsidence is defined as the lowering or collapse of the land surface either locally or over broad regional areas (USGS, 1981). Subsidence can be the result of either natural phenomena or human activities.

Natural causes of subsidence include disolution of limestone and other soluble materials, earthquakes, and volcanic activity. Limestone and dolomite are slightly soluble in water and, especially in hot and wet climates, excessive moisture can cause voids to form in their mineral formations. If overlying materials collapse or subside into the solution cavities, a surface depression called a sinkhole will form. Where limestone or dolomite deposits are widespread and sinkholes are common, the land surface is referred to as karst topography or karst terrains after a district in Yugoslavia where the phenomenon is common (Bolt, 1975).

Sinkholes vary in depth from slight indentations to depressions over 100-ft deep. Typically, depths range from 10-30 ft with areas ranging from a few square yards to several acres. There are two major classes of sinkholes: those forming from the collapse of an underground void, and the more common type, the doline, that develops slowly by the downward solution of the underlying rock. Surface water entering the first type of sinkhole tends to flow rapidly underground through outlets called swallow holes. Runoff draining into a doline, however, usually percolates slowly underground through the soil in the bottom of the sinkhole. If a sinkhole becomes clogged, a temporary pond or lake will form; if the outlet is unplugged, rapid drainage may occur. This phenomenon occurred at Lake Jackson, a lake of approximately 25 square kilometers near Tallahassee, Florida, that drained on May 22, 1982.

Karst terrains and caverns in the United States are found primarily in parts of many southeastern and midwestern States. Sinkholes are found in some western and northeastern States and in Alabama, where soluble limestone and other rocks present in nearly half of the State have created thousands of sinkholes that pose serious problems for highways and construction.

Earthquake-related subsidence has taken place mainly in Alaska, California, and Hawaii, and to a lesser extent in other States. This type of subsidence results from the vertical movement of faults and may effect broad areas. In 1964 in southern Alaska, in conjunction with the Prince William Sound, Alaska, earthquake, more than 70,000 square miles tilted downward more than 3 ft and subsequently flooded. Subsidence resulting from intense earthquake groundshaking involves somewhat smaller areas than that resulting from regional vertical faulting. Intense groundshaking generated during earthquakes in 1811-1812 in New Madrid, Missouri, caused subsurface sand and water to be ejected to the surface. This ejection left voids in the subsurface, causing local compaction of subsurface materials and settling of the ground.

Volcanic-related subsidence is a potential problem in parts of Alaska, California, Hawaii, Oregon, and Washington. Subsidence usually is caused by local collapses above shallow tunnels formed by lava flows. Collapses over much broader areas also can occur as magma chambers are emptied by volcanic eruptions (USGS, 1981).

Human activities that cause subsidence include the withdrawal of water, oil, or gas from the pore spaces in subsurface formations. When these fluids are removed by pumping, the effective stress on the formation increases as the pore pressure decreases. This stress can cause the formation to compress vertically, resulting in land-surface subsidence.

The most dramatic examples of subsidence caused by withdrawal of oil, gas, and water are along the Gulf Coast of Texas, in Arizona, and in California. The harbor at Long Beach, California, has subsided as much as 27 ft from withdrawal of gas and oil. The Houston-Galveston area of Texas has experienced as much as 7.5 ft of subsidence locally and an area of about 2,500 square miles has subsided 1 ft or more. Subsidence in the Houston-Galveston area appears to have been caused mainly by the withdrawal of large amounts of ground water, although some areas of local subsidence have been caused by the extraction of gas and oil. Coastal towns in Texas, such as Baytown and Seabrook, have subsided about 3 ft and are now susceptible to flooding from storm surges and hurricanes (USGS, 1981).

Recent research suggests that subsidence caused by withdrawing ground water can cause fissuring (the formation of open cracks) or renewal of surface movement in some areas cut by pre-existing faults. Surface faulting and fissuring associated with withdrawal of ground water are believed to have taken place or to be potential problems in the vicinity of Las Vegas, Nevada, as well as in parts of Arizona, California, Texas, and New Mexico (USGS, 1981).

The removal of solid materials from underground deposits can also result in subsidence. Subsidence has occurred in shallow coal mines where a lack of adequate roof support caused collapses either during mining or long after mining had ceased. Subsidence in areas of underground mining has caused hazardous conditions in parts of Pennsylvania and other Appalachian States, Colorado, North Dakota, Wyoming, New Mexico, Washington, Iowa, and Illinois. Subsidence-related damage to surface structures is common in the area around Pittsburgh, Pennsylvania, where coal has been mined extensively. Subsidence depressions and pits forming above abandoned underground mines are a hazard in the Sheridan, Wyoming, area.

Solution mining also can cause subsidence. When water-soluble minerals such as salt, gypsum, and potash are dissolved and pumped to the surface so that the water can be evaporated, huge underground cavities are formed. Examples of subsidence that are not well known, include the sudden collapse in 1976 of a Grand Saline, Texas street into an abandoned salt mine cavity created between 1924 and 1949; subsidence in 1974 near Hutchinson, Kansas; and subsidence in 1971 near Detroit, Michigan (USGS, 1981).

The addition of water to some soils and soft rock formations can cause subsidence. One type of formation where this can happen is called loess. Loess has the characteristics of soft rock, being formed from fine-grained material cemented together in an extremely loose, open structure by a water-soluble mineral cement. In the dry state, loess is firm and hard and serves well as a foundation material. When subjected to excessive wetting from sprinkling or ponding on the surface, however, the mineral cement dissolves and the soil structure collapses in the saturated area.

Hydrocompaction, or the settling of sediments after water is added, is another significant cause of subsidence, especially in the arid to semiarid western and midwestern States. The areas of known compaction include the San Joaquin Valley, California; Heart Mountain-Chapman Beach and Riverton, Wyoming areas; Hysham, Montana; Columbia Basin, Washington; Denver, Colorado; Washington-Hurricane area in southwest Utah and central Utah; and the Missouri River Basin. Hydrocompaction takes place when dry surface or subsurface deposits are wet extensively for the first time since their deposition as, for example, when arid land is irrigated for crop production or an irrigation canal is built on loose uncompacted sediments. Wetting causes a reduction in the cohesion between sediment grains, allowing the grains to fill in the naturally occurring intergranular openings. Hydrocompaction usually results in a lowering of the land surface from 3-6 ft, although subsidence of as much as 15 ft has been recorded. The effects of hydrocompaction are usually uneven, causing depressions, cracks, and wavy surfaces that can seriously damage canals, highways, pipelines, buildings, and other structures (USGS, 1981).

6.6.2 Impacts of Siting Facilities in Unstable Areas

In landslide-prone areas, construction of monofills on natural slopes can cause problems if the shear strength of underlying materials is exceeded by the placement of waste overburden. Vibrations generated by construction equipment and heavy vehicular traffic on the site could also trigger ground movement. During liner construction, care must be taken not to remove the toe of adjacent slopes that could slide into the monofill.

In areas with expansive clay, liners can be torn or rocked and sidewalls can collapse if uneven expansion occurs below the monofill. To avoid this situation, expansive soils can be treated in situ, removed from the site, or sealed to prevent changes in their moisture content. If expansive clay soils are deep enough, sufficient overburden can be left in place to prevent excessive swelling. Regardless of the solution, problems must be recognized early in the planning stage in order to include proper expansive soil control measures in the facility design.

In areas that are plagued with subsidence problems due to underground cavities, there is always a possibility of subsidence occurring below a monofill. This is a particular hazard if the weight of the placed waste exceeds the bearing strength of the materials supporting the facility.

The Agency examined case studies and documented events that cited the types of environmental problems resulting from locating waste management units in unstable areas. Differential or total foundation movement, loss of facility structural integrity, sudden and/or emergency release of contaminants, difficult and costly corrective actions, and contaminant releases attributed to maintenance or inadequate facility design are all hazards of locating monofills in unstable areas. The Agency believes that it is essential for the owner or operator to evaluate extensively the adequacy of subsurface foundation support before constructing a monofill.

6.6.3 Alternatives

The Agency considered three alternative approaches for regulating the siting of monofills in unstable areas:

1) Provide no standard relating to the siting of monofills in unstable areas.
2) Develop detailed maps of all unstable areas and prohibit siting of monofills in those areas.
3) Require demonstrations by all owners/operators that a site is not subject to damage from unstable conditions.

The first alternative was the simplest and least costly to implement; however, it was the least protective of the environment and human health.

The second alternative, to map all unstable areas and ban monofills from these areas, was clearly unworkable. Many unstable areas cover large geographic regions and to ban monofills within these areas would impose a great economic burden on communities located in those areas. Additionally the task of mapping all unstable areas with sufficient detail and accuracy probably is beyond the capabilities of modern geoscience, as well as too costly to receive serious consideration.

The third alternative provides the best approach to controlling the siting of monofills. Owners/operators would be required to perform geotechnical studies of all proposed sites in order to demonstrate that unstable conditions do not exist. This method allows the placement of facilities within broad geographic areas where unstable conditions may potentially exist while at the same time prohibits the placement of a facility on a specific site where unstable conditions actually do exist.

6.6.4 Regulatory Requirement for Siting Monofills in Unstable Areas

The proposed standard requires that sewage sludge units be located in areas where adequate support for the structural components of the units exist. Although unstable support conditions may be common over large areas of the country, there may be places within these regions where monofills can be safely sited.

For the most part, identification of monofill sites in unstable areas must be on a case-by-case basis because these areas have not been delineated on a national scale. National maps are available describing karst terrains and landslide-susceptible areas, but weak and unstable soils and subsidence-prone areas appear to be mapped only individually or at the local level.

Under the proposed requirement, the facility owner or operator must determine if an area is subject to events that may cause or contribute to the facility's failure. If the unit is located in an unstable area, the owner or operator must demonstrate to the State that the proposed site is not subject to destabilizing events.

SECTION SEVEN

MANAGEMENT PRACTICES

Six areas of management practice requirements are specified by the proposed sewage sludge regulations. One of these six, pathogen reduction, is covered under separate title (EPA, 1989a). Of the remaining five, landfill cover requirements are not relevant to surface disposal. Runon/runoff controls are also not covered here because surface disposal units are expected to be quite small. Disease vector controls, explosive gases controls, and access controls are discussed in this section. The growing of food and feed crops on surface disposal units as well as animal grazing will also be prohibited. These management practices are required under Subtitle D or C of the Resource Conservation and Recovery Act (RCRA) and have been incorporated to some extent into the proposed sewage sludge regulation. The way in which the requirements have been incorporated and the rationale for the approach used to incorporate or specify alternatives to these requirements are presented below for each of the management practices covered in this document.

7.1 DISEASE VECTOR CONTROL REQUIREMENTS

Surface disposal units can provide food, shelter, and breeding areas for disease vectors, which are animals such as rats, flies, and mosquitoes that can transmit disease to humans. Some diseases associated with these vectors are rat-bite fever, leptospirosis, plague, salmonellosis, trichinosis, murine/typhus fever, malaria, and yellow fever (Noble, 1976; EPA, 1979) Because surface disposal units can provide an environment for a large number of disease vectors, EPA requires that owners or operators conscientiously monitor the disease vector population and follow practices to minimize that population. EPA further requires surface disposal unit owners or operators to employ prevention and control techniques because accurately measuring specific levels of disease vectors would be difficult or impossible.

Judiciously applied poisons (insecticides and rodenticides) are an effective control measure. Insecticide sprays can be used to control fly and mosquito populations, and rodenticides can be used to control rats. Rodenticides, however, serve strictly as short-term solutions. It should be noted that insecticides or rodenticides must be used according to label instructions to avoid adverse environmental impacts.

Predatory or reproductive controls can help minimize the vector population. For example, organisms that feed on mosquito larvae could be introduced to the facility. Agitating and/or varying the level of standing water (e.g., leachate ditches) also helps prevent mosquito reproduction. (EPA, 1980).

7.2 REQUIREMENTS FOR CONTROL OF EXPLOSIVE GASES

When organic matter in monofills is decomposed by anaerobic bacteria, explosive gases, predominantly methane, are produced. This process also occurs in surface disposal sites. Anaerobic conditions arise when the conditions in the voids within the waste materials change from aerobic to anaerobic and the chemically available oxygen in the refuse is consumed. The process of gas production is controlled by site-specific conditions that affect the bacterial population, such as pH, temperature, moisture, and oxygen content (both gaseous and chemically available). The following discussion about landfills and landfill gases holds true for surface disposal sites.

As landfill gas is generated, pressure builds up within the material. The pressure then forces the gas to migrate laterally and/or vent to the atmosphere. The two basic mechanisms by which gases migrate are molecular diffusion, which is slow, and convective mass transfer, or pressure-induced flow, which is the predominant mechanism of subsurface gas flow. (GCA, 1986c).

The methane component of monofill gas can create an explosion hazard. Migrating methane has forced the evacuation of residences and businesses, and has been responsible for onsite and offsite fires, explosions, property damage, human injury, and in some cases, death (GCA, 1986c). In addition to property damage, groundwater contamination and vegetative destruction on site and on adjacent lands have resulted from landfill gas migration (EPA, 1977a).

For these reasons, EPA established an explosive gas criterion in the original Subtitle D Criteria under 40 CFR 257.3-8(a) that sets limits on explosive gases (methane) in facility structures and at the property boundaries. This standard requires that concentrations of methane not exceed the lower explosive level (LEL) (5% methane) in soil at the property boundary and 25% of the LEL (1.25% methane) in onsite structures. Because the human health standard for methane is greater than the LEL, EPA believes the current limits provide adequate protection to human health and the environment at the property boundary and allow for a margin of safety for onsite structures. In selecting the 25% figure, EPA is using a safety factor recognized as appropriate for similar structures by other Federal agencies, e.g., in regulations established by the Occupational Safety and Health Administration and the National Fire Protection Codes.

The final method selected to prevent or control the formation and migration of monofill gases will depend on site-specific conditions, including soil type, geology, hydrology, and climatic conditions. Numerous methods that have been used to control gas migration include (EPA, 1977; Raymond Vail and Associates, 1979):

- Permeable trench (gravel trench)
- Semipermeable barrier (synthetic liner or clay liner) used with a gravel trench (trench barrier system)
- Venting system with gravel trench or gravel layers (passive landfill control systems)
- Venting with a gas pumping system (active landfill vent control systems)

A report prepared by Raymond Vail and Associates (1979) provides more detail on the gas migration control systems discussed above.

A permeable trench that collects and vents the gas is placed at the perimeter of a site where gas migration is of most concern. The trench generally is installed at a depth equal to the depth of the site, but this approach is not the best approach, because gas may migrate under the trench. Ideally the trench should be installed to ground water depth or to the depth of an impermeable geological barrier. The permeable-trench method of venting, however, has had a low percentage of success in controlling gas migration (EPA, 1977a; Raymond Vail and Associates, 1979).

The second method, a gravel trench used with a semipermeable liner (a trench barrier system), provides a more effective barrier than the permeable trench system. The trench-barrier system should be installed to water table depth and care must be taken during installation of the liner material to avoid tears (EPA, 1977a; Raymond Vail and Associates, 1979).

The third method, a pipe venting system used with a gravel trench (a passive landfill vent control system), has approximately the same effectiveness for controlling gas migration as trenches used alone. In addition, this system is more costly than the above two methods and may be subject to vandalism (EPA, 1977).

The last method, the pumping system or active landfill vent control system, provides the greatest effectiveness for gas migration control but is more costly to install and maintain. This system incorporates a series of wells (not necessarily installed to the water table) that withdraw the gas from the landfill or from the area surrounding the landfill. Generally, if recovered gas is to be used as an energy source, an active vent system is used. With this method, recovery and migration control are accomplished in a single process. Where gas migration is to be controlled, pumping wells are installed between the source of the gas and the threatened building or home.

To avoid continuous operation, systems may be designed to operate only when a certain concentration of methane is detected (EPA, 1977).

7.3 ACCESS CONTROL REQUIREMENTS

Controlling access to surface disposal sites is necessary to prevent injury to the public from facility hazards and to ensure that all wastes are properly disposed. Some sources of potential injury at both landfills and surface disposal sites are (EPA, 1980):

- Operations involving heavy equipment and haul vehicles
- Waste related hazards; such as pathogens
- Accidental or intentional fires
- Excavations and other earth moving activities

The potential harm that these hazards pose to facility personnel can be controlled through proper training, use of safety equipment, and other procedures (44 FR 53460, 1979). These controls, however, cannot be imposed on the general public. The only way to prevent potential harm to the public is by prohibiting access to surface disposal sites by the general public and by strictly controlling users of the facility on the site (44 FR 53460, 1979).

Control of authorized persons at surface disposal sites can be achieved using any of the following measures (EPA, 1980):

- Supervising the unloading area
- Providing adequate lighting
- Posting information and direction signs
- Prohibiting scavenging

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